Urban Stormwater Management in the United States

Committee on Reducing Stormwater Discharge Contributions to Water Pollution, National Research Council

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3
Hydrologic, Geomorphic, and Biological Effects of Urbanization on Watersheds

A watershed is defined as the contributing drainage area connected to an outlet or waterbody of interest, for example a stream or river reach, lake, reservoir, or estuary. Watershed structure and composition include both naturally formed and constructed drainage networks, and both undisturbed areas and human dominated landscape elements. Therefore, the watershed is a natural geographic unit to address the cumulative impacts of urban stormwater. Urbanization has affected change to natural systems that tends to occur in the following sequence. First, land use and land cover are altered as vegetation and topsoil are removed to make way for agriculture or subsequently buildings, roads, and other urban infrastructure. These changes, and the introduction of a built drainage network, alter the hydrology of the local area, such that receiving waters in the affected watershed can experience radically different flow regimes than they did prior to urbanization. This altered hydrology, when combined with the introduction of pollutant sources that accompany urbanization (such as people, domesticated animals, industries, etc.), has led to water quality degradation of many urban streams.

This chapter first discusses the typical land-use and land-cover composition of urbanized watersheds. This is followed by a description of changes to the hydrologic and geomorphic framework of the watershed that result from urbanization, including altered runoff, streamflow mass transport, and stream-channel stability. The chapter then discusses the characteristics of stormwater runoff, including its quantity and quality from different land covers, as well as the characteristics of dry weather runoff. Finally, the effects of urbanization on aquatic ecosystems and human health are explored.

LAND-USE CHANGES

Land use has been described as the human modification of the natural environment into the built environment, such as fields, pastures, and settlements. Important characteristics of different land uses are the modified surface characteristics of the land and the activities that take place within that land use. From a stormwater viewpoint, land uses are usually differentiated by building density and comprised of residential, commercial, industrial, institutional, recreational, and open-space land uses, among others. Each of these land uses usually has distinct activities taking place within it that affect runoff quality. In addition, each land use is comprised of various amounts of surface land cover, such as roofs, roads, parking areas, and landscaped areas. The amount and type of each cover also affect the quality and quantity of runoff from urban areas. Changes in land use and in the land covers within the land uses associated with develop-
ment and redevelopment are therefore important considerations when studying local receiving water problems, the sources of these problems within the watershed, and the stormwater control opportunities.

**Land-Use Definitions**

Although there can be many classifications of residential land use, a crude and common categorization is to differentiate by density. High-density residential land use refers to urban single-family housing at a density of greater than 6 units per acre, including the house, driveway, yards, sidewalks, and streets. Medium density is between 2 and 6 units per acre, while low density refers to areas where the density is 0.7 to 2 units per acre. Another significant residential land use is multiple-family housing for three or more families and from one to three stories in height. These units may be adjoined up-and-down, side-by-side, or front-and-rear.

There are a variety of commercial land uses common in the United States. The strip commercial area includes those buildings for which the primary function is the sale of goods or services. This category includes some institutional lands found in commercial strips, such as post offices, court houses, and fire and police stations. This category does not include warehouses or buildings used for the manufacture of goods. Shopping centers are another common commercial area and have the unique distinction that the related parking lot that surrounds the buildings is at least 2.5 times the area of the building roof area. Office parks are a land use on which non-retail business takes place. The buildings are usually multi-storied and surrounded by larger areas of lawn and other landscaping. Finally, downtown central business districts are highly impervious areas of commercial and institutional land use.

Industrial areas can be differentiated by the intensity of the industry. For example, “manufacturing industrial” is a land use that encompasses those buildings and premises that are devoted to the manufacture of products, with many of the operations conducted outside, such as power plants, steel mills, and cement plants. Institutional areas include a variety of buildings, for example schools, churches, and hospitals and other medical facilities that provide patient overnight care.

Roads constitute a very important land use in terms of pollutant contributions. The “freeway” land use includes limited-access highways and the interchange areas, including any vegetated rights-of-ways. Finally, there are a variety of open-space categories, such as cemeteries, parks, and undeveloped land. Parks include outdoor recreational areas such as municipal playgrounds, botanical gardens, arboretums, golf courses, and natural areas. Undeveloped lands are private or publicly owned with no structures and have a complete vegetative cover. This includes vacant lots, transformer stations, radio and TV transmission areas, water towers, and railroad rights-of-way.

The preceding land-use descriptions are the traditional categories that make
up the vast majority of the land in U.S. cities. However, there are emerging categories of land use, such as those espoused under the term New Urbanism, which combine several area types (such as commercial and high-density residential areas). Although land use can be broadly and generally categorized, local variations can be extremely important such that locally available land-use data and definitions should always be used. For example, local planning agencies typically do not separate the medium-density residential areas into subcategories. However, this may be necessary to represent different development trends that have occurred with time, and to represent newly emerging types of land uses for an area. Box 3-1 discusses the subtle influence that tree canopy could have on the residential land-use classification.

**Trends in Urbanization**

Researchers at Columbia University (de Sherbinin, 2002) state that 83 percent of the Earth’s land surface has been affected by human settlements and activities, with the urbanized areas comprising about 4 percent of the total land use of the world. Urban areas are expanding world-wide, especially in developing countries. The United Nations Population Division estimates suggest that the

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**BOX 3-1**

**The Role of Tree Cover in Residential Land Use**

Figure 3-1 shows two medium-density residential neighborhoods, one older and one newer. Tree canopy is obviously different in each case, and it may have an effect on seasonal organic debris in an area and possibly on nutrient loads (although nutrient discharges appear to be more related to homeowner fertilizer applications). Increased tree canopy cover also has a theoretical benefit in reducing runoff quantities due to increased interception losses. In both cases, however, monitoring data to quantify these benefits are sparse. Xiao (1998) examined the effect urban tree cover had on the rainfall volume striking the ground in Sacramento, California. The results indicated that the type of tree or type of canopy cover affected the amount of rainfall reduction measured during a rain event, such that large broad-leaved evergreens and conifers reduced the rainfall that reached the ground by 36 percent, while medium-sized conifers and deciduous trees reduced the rainfall by 18 percent. Cochran (2008) compared the volume and intensity of rain that reached the ground in an open area (no canopy cover) versus two areas with intact canopy covers in Shelby County, Alabama, over a year. The sites were sufficiently close to each other to assume that the rainfall characteristics were the same in terms of the intensity and the variation of intensity and volume during the storm. Rainfall “throughfall” was reduced by about 13.5 percent during the spring and summer months when heavily wooded cover existed. The rainfall characteristics at the leafless tree sites (winter deciduous trees) were not significantly different from the parking lot control sites. In many locations around the county, very high winds are associated with severe storms, significantly decreasing the interception losses. Of course, mature trees are known to provide other benefits in urban areas, including shading to counteract stormwater temperature increases and massive root systems that help restore beneficial soil structure conditions. Additional research is needed to quantify the benefits of urban trees through a comprehensive monitoring program.

*continues next page*
BOX 3-1 Continued

FIGURE 3-1 Two medium-density residential areas (no alleys); the area below is older. SOURCE: Robert Pitt, University of Alabama.
world’s population will become mostly urbanized by 2010, whereas only 37 percent of the world’s population was urbanized in 1970. De Sherbinin (2002) concludes that although the extent of urban areas is not large when compared with other land uses (such as agriculture or forestry) their environmental impact is significant. Population densities in the cities are large, and their political, cultural, and economic influence is great. Most industrial activity is also located near cities. The influence of urban areas extends beyond their boundaries due to the need for large amounts of land for food and energy production, to generate raw materials for industry, for building water supplies, for obtaining other resources such as construction materials, and for recreational areas. One study estimated that the cities of Baltic Europe require from 500 to more than 1,000 times the urbanized land area (in the form of forests, agricultural, marine, and wetland areas) to supply their resources and to provide for waste disposal (de Sherbinin, 2002).

Currently, considerable effort is being spent investigating land-use changes world-wide and in the United States in support of global climate change research. The U.S. Geological Survey (USGS, 1999) has prepared many research reports describing these changes; Figure 3-2 shows the results for one study in the Chicago and Milwaukee areas, and Figure 3-3 shows the results for a study in the Chesapeake Bay area. These maps graphically show the dramatic rate of change in land use in these areas. The very large growth in urban areas during the 20 years between 1975 and 1995 is especially astonishing. By 1995, Milwaukee and Chicago’s urbanized areas more than doubled in size from prior years. Even more rapid growth has occurred in the Washington, D.C.–Baltimore area.

FIGURE 3-3  This series of maps compares changes in urban, agricultural, and forested lands in the Patuxent River watershed over the past 140 years. The top series shows the extent of urban areas (black) along with agriculture (gray), which was at its peak in the mid- to late 1800s. The bottom series shows that urban (black) and forested land (gray) have increased since 1900. SOURCE: USGS (1999).
Many different metrics can be used to measure the rate of urbanization in the United States, including the number of housing starts and permits and the level of new U.S. development. The latter is tracked by the U.S. Department of Agriculture’s (USDA) National Resources Inventory (USDA, 2000). The inventory, conducted every five years, covers all non-federal lands in the United States, which is 75 percent of the U.S. total land area. The inventory uses land-use information from about 800,000 statistically selected locations. From 1992 to 1997, about 2.2 million acres per year were converted from non-developed to developed status. According to the USDA (2000), the per capita developed land use (acres per person, a classical measure of urban sprawl) has increased in the United States between the years of 1982 and 1997 from about 0.43 to about 0.49 acres per person. The smallest amount of developed land used per person was for New York and Hawaii (0.15 acres), while the largest land consumption rate was for North Dakota, at about 10 times greater. Surprisingly, Los Angeles is the densest urban area in the country at 0.11 acres per person. The amount of urban sprawl is also directly proportionate to the population growth. According to Beck et al. (2003):

In the 16 cities that grew in population by 10 percent or less between 1970 and 1990 (but whose population did not decline), developed area expanded 38 percent—more than in cities that declined in population but considerably less than in the cities where population increased more dramatically. Cities that grew in population by between 10 and 30 percent sprawled 54 percent on average. Cities that grew between 31 and 50 percent sprawled 72 percent on average. Cities that grew in population by more than 50 percent sprawled on average 112 percent. These findings confirm the common sense, but often unacknowledged proposition, that there is a strong positive relationship between sprawl and population growth.

In most areas, the per capita use of developed land has increased, along with the population growth. However, even some cities that had no population growth or had negative growth, such as Detroit, still had large amounts of sprawl (increased amounts of developed land used per person), but usually much less than cities that had large population growth. Los Angeles actually had an 8 percent decreased rate of land consumption per resident during this period, but the city still experienced tremendous growth in land area due to its very large population growth. The additional 3.1 million residents in the Los Angeles area during this time resulted in the development of almost an additional 400 square miles.

**Land-Cover Characteristics in Urban Areas**

As an area urbanizes, the land cover changes from pre-existing rural sur-
faces, such as agricultural fields or forests, to a combination of different surface types. In municipal areas, land cover can be separated into various common categories—pictured and described in Box 3-2—that include roofs, roads, parking areas, storage areas, other paved areas, and landscaped or undeveloped areas.

Most attention is given to impervious cover, which can be easily quantified for different types of land development. Given the many types of land cover described in Box 3-2, impervious cover is composed of two principal components: building rooftops and the transportation system (roads, driveways, and parking lots). Compacted soils and unpaved parking areas and driveways also have “impervious” characteristics in that they severely hinder the infiltration of water, although they are not composed of pavement or roofing material. In terms of total impervious area, the transportation component often exceeds the rooftop component (Schueler, 1994). For example, in Olympia, Washington, where 11 residential multifamily and commercial areas were analyzed in detail, the areas associated with transportation-related uses comprised 63 to 70 percent of the total impervious cover (Wells, 1995). A significant portion of these impervious areas—mainly parking lots, driveways, and road shoulders—experience only minimal traffic activity. Most retail parking lots are sized to accommodate peak parking usage, which occurs only occasionally during the peak holiday shopping season, leaving most of the area unused for a majority of the time. On the other hand, many business and school parking areas are used to their full capacity nearly every work day and during the school year. Other differences at parking areas relate to the turnover of parking during the day. Parked vehicles in business and school lots are mostly stationary throughout the work and school hours. The lighter traffic in these areas results in less vehicle-associated pollutant deposition and less surface wear in comparison to the greater parking turnover and larger traffic volumes in retail areas (Brattebo and Booth, 2003).

As described in Box 1-1, impervious cover is broken down into two main categories: directly connected impervious areas (or effective impervious area) and non-directly connected (disconnected) impervious areas (Sutherland, 2000; Gregory et al., 2005) (although it is recognized that these two states are end-members of a range of conditions). Directly connected impervious area includes impervious surfaces which drain directly to the sealed drainage system without flowing appreciable distances over pervious surfaces (usually a flow length of less than 5 to 20 feet over pervious surfaces, depending on soil and slope characteristics and the amount of runoff). Those areas are the most important component of stormwater runoff quantity and quality problems. Approximately 80 percent of directly connected impervious areas are associated with vehicle use such as streets, driveways, and parking (Heaney, 2000).

Values of imperviousness can vary significantly according to the method used to estimate the impervious cover. In a detailed analysis of urban imperviousness in Boulder, Colorado, Lee and Heaney (2003) found that hydrologic modeling of the study area resulted in large variations (265 percent difference)
For any given land use, there is a range of land covers that are typical. Common land covers are described below, along with some indication of their contribution to stormwater runoff and their pollutant-generating ability.

**Roofs.** These are usually either flat or pitched, as both have significantly different runoff responses. Flat roofs can have about 5 to 10 mm of detention storage while pitched roofs have very little detention storage. Roofing materials are also usually quite different for these types of roofs, further affecting runoff quality. In addition, roof flashing and roof gutters may be major sources of heavy metals if made of galvanized metal or copper. Directly connected roofs have their roof drains efficiently connected to the drainage system, such as direct connections to the storm drainage itself or draining to driveways that lead to the drainage system. These directly connected roofs have much more of their runoff waters reaching the receiving waters than do partially connected roofs, which drain to pervious areas.

**Parking Areas.** These can be asphalt or concrete paved (impervious surface) or unpaved (traditionally considered a pervious surface) and are either directly connected or drain to adjacent pervious areas. Areas that have rapid turnover of parked cars throughout the day likely have greater levels of contamination due to the frequent starting of the vehicles, an expected major source of pavement pollutants. Unpaved parking areas actually should be considered impervious surfaces, as the compacted surface does not allow any infiltration of runoff. Besides automobile activity in the parking areas, other associated activities contribute to contamination. For example, parked cars in disrepair awaiting service can contribute to parking area runoff contamination. In addition, maintenance of the pavement surface, such as coal-tar seal coating, can be significant sources of polycyclic aromatic hydrocarbons (PAHs) to the runoff.

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_A directly connected roof drain_  
_A disconnected roof drain (drains to pervious area)_

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Storage Areas. These can also be paved, unpaved, directly connected, or drained to pervious areas. As with parking areas, unpaved storage areas should not be considered pervious surfaces because the compacted material effectively hinders infiltration. Detention storage runoff losses from unpaved storage areas can be significant. In storage areas (especially in commercial and industrial land uses), activities in the area can have significant effects on runoff quality.

Streets. Streets in municipal areas are usually paved and directly connected to the storm drainage system. In municipal areas, streets constitute a significant percentage of all impervious surfaces and runoff flows. Features that affect the quality of runoff from streets include the varying amounts of traffic on different roads and the amount and type of roadside vegetation. Large seasonal phosphorus loads can occur from residential roads in heavily wooded areas, for example.
Other Paved Areas. Other paved areas in municipal regions include driveways, playgrounds, and sidewalks. Depending on their slopes and local grading, these areas may drain directly to the drainage system or to adjacent pervious areas. In most cases, the runoff from these areas contributes little to the overall runoff for an area, and the runoff quality is of relatively better quality than from the other "hard" surfaces.

Landscaped and Turf Areas. Although these are some of the only true pervious surfaces in municipal areas, disturbed urban soils can be severely compacted, with much more reduced infiltration rates than are assumed for undisturbed regional soils. Besides the usually greater than expected quantities of runoff of pervious surfaces in urban areas, they can also contribute high concentrations of various pollutants. In areas with high rain intensities; erosion of sediment can be high from pervious areas, resulting in much higher concentrations of total suspended solids (TSS) than from paved areas. Also, landscaping chemicals, including fertilizers and pesticides, can be transported from landscaped urban areas. Undeveloped woods in urban areas can have close to natural runoff conditions, but many parks and other open-space areas usually have degraded runoff compared to natural conditions. Turf grass has unique characteristics compared to other landscaped areas in that the soil structure is usually more severely degraded compared to natural conditions. The normally shallower root systems are not as effective in restoring compacted soils and they can remain compacted due to some activities (pathways, parked cars, playing fields, etc.) that do not occur on areas planted with shrubs and trees.

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Undeveloped Areas. Undeveloped areas in otherwise urban locations differ from natural areas. In many situations, they can be previously disturbed (cleared and graded) areas that have not been sold or developed. They may be overgrown with various local vegetation types that thrive in disturbed locations. In other situations, undeveloped areas may be small segments of natural areas that have not been disturbed or revegetated. In this case, their stormwater characteristics may approach natural conditions but still be degraded due to adjacent activities and atmospheric deposition.


in the calculations of peak discharge when impervious surface areas were determined using different methods. They concluded that the main focus should be on effective impervious area (EIA) when examining the effects of urbanization on stormwater quantity and quality.

Runoff from disconnected impervious areas can be spread over pervious surfaces as sheet flow and given the opportunity to infiltrate before reaching the drainage system. Therefore, there can be a substantial reduction in the runoff volume and a delay in the remaining runoff entering the storm drainage collection system, depending on the soil infiltration rate, the depth of the flow, and the
available flow length. Examples of disconnected impervious surfaces are rooftops that discharge into lawns, streets with swales, and parking lots with runoff directed to adjacent open space or swales. From a hydrologic point of view, road-related imperviousness usually exerts a larger impact than rooftop-related imperviousness, because roadways are usually directly connected whereas roofs can be disconnected (Schueler, 1994).

**Methods for Determining Land Use and Land Cover**

Historically, land-use and land-cover information was acquired by a combination of field measurements and aerial photographic analyses—methods that required intensive interpretation and cross validation to guarantee that the analyst’s interpretations were reliable (Goetz et al., 2003). Figure 3-4 is an example of a high-resolution panchromatic aerial photograph that was taken from an airplane in Toronto and used for measurements of urban surfaces (Pitt and McLean, 1986). Most recently, satellite images have become available at high spatial resolution for many areas (<1 to 5 m resolution) and have the advantage of digital multi-spectral information more complete than even that provided by digital orthophotographs. Minnesota has one of the longest records (over 20 years) of continuously recorded statistics on land cover and impervious surfaces derived from satellite images—information which has been incorporated into the

![Figure 3-4](image-url)

**FIGURE 3-4** Example of a high-resolution panchromatic aerial photograph of an industrial area used for measurements of urban surfaces. **SOURCE:** Pitt and McLean (1986).
Minnesota Statewide Conservation and Preservation Plan. Some of the remaining problems to be overcome with satellite imagery include difficulties in obtaining consistent sequential acquisition dates, intensive computer processing time requirements, and large computer storage space requirements to store massive amounts of image information.

The recommended approach for conducting a survey of land uses and development characteristics (land cover and activities) for an area is to use both aerial photography and site surveys. Aerial photography has improved greatly in recent years, but it is still not suitable for obtaining all the information needed for developing a comprehensive stormwater management plan. Initially, aerial photos should be used to identify the locations and extents of the various land uses in the study area. Neighborhoods representing homogenous land uses should then be identified for site surveys. Usually, about 10 to 15 neighborhoods for each land use are sufficient for a community being studied (Burton and Pitt, 2002). After the field surveys are conducted, the aerials are again used to measure the actual areas associated with land surface cover. This information can be used with field survey data to separate the surfaces into the appropriate categories for analyses and modeling.

Box 3-3 presents a detailed study of land cover for several land uses in the southern United States using satellite imagery and ground surveys (Bochis, 2007; Bochis et al., 2008). The results presented here have been found to be broadly similar to other areas studied in the United States, although few studies have been as detailed, and there are likely to be regional differences.

The general conclusion of many land-use and land-cover studies is that in urban areas, the amount of impervious surfaces has increased since the early years of the 20th century because of the tendency toward increased automobile use and bigger houses, which is associated with an increase in the facilities necessary to accommodate them (wider streets, more parking lots, and garages). As shown in later sections of this report, the construction of impervious surfaces leads to multiple impacts on stream systems. Therefore, future development plans and water resource protection programs should consider reducing impervious cover in the potential expansion of communities. Wells (1995), Booth (2000), Stone (2004), and Gregory et al. (2005) show that reducing the size and dimensions of residential parcels, promoting cluster developments (clustered medium-density residential areas in conjunction with open space, instead of large tracts of low-density areas), building taller buildings, reducing the residential street width (local access streets), narrowing the width and/or building one-side sidewalks, reducing the size of paved parking areas to reflect the average parking needs instead of peak needs, and using permeable pavement for intermittent/overflow parking can reduce the traditional impervious cover in communities by 10 to 50 percent. Many of these benefits can also be met by paying better attention to how the pavement and roof areas are connected to the drainage system. Impervious surfaces that are “disconnected” by allowing their drainage water to flow to adjacent landscaped areas can result in reduced runoff quantities.
Land Use and Land Cover for the Little Shades Creek Watershed

Data collected by Bochis-Micu and Pitt (2005) and Bochis (2007) for the Little Shades Creek watershed near Birmingham, Alabama, were acquired using IKONOS satellite imagery (provided by the Jefferson County Storm Water Management Authority) as an alternative to classical aerial photography to map the characteristics of the land uses in the monitored watershed areas, supplemented with verified ground truth surveys. IKONOS is the first commercially owned satellite that provides 1-m-resolution panchromatic image data and 4-m multi-spectral imagery (Goetz et al., 2003).

This project was conducted to evaluate the effects of variable site conditions associated with each land-use category. About 12 homogeneous neighborhoods were investigated in each of the 16 major land uses in this 2,500-hectare watershed. Detailed land-cover measurements were made using a variety of techniques, as listed above, including field surveys for small details that were not visible with remote sensing tools (such as roof drain connectiveness, pavement texture, and landscaping maintenance practices). Each of these individual neighborhoods was individually modeled to investigate the resultant variability in runoff volume and pollutant discharges. These were statistically evaluated to determine if the land-use categories properly stratified these data by explaining significant fractions of the variability. Bochis-Micu and Pitt (2005) and Bochis (2007) concluded that land-use categories were an appropriate surrogate that can be used to describe the observed combinations of land surfaces. However, proper stormwater modeling should examine the specific land surfaces in each land-use category in order to better understand the likely sources of the pollutants and the effectiveness of candidate stormwater control measures (SCMs).

This watershed has an overall impervious cover of about 35 percent, of which about 25 percent is directly connected to the drainage system. Table 3-1 shows the average land covers for each of the surveyed land uses, along with the major source areas in each of the directly connected and disconnected impervious and pervious surface categories. The impervious covers include streets, driveways, parking, playgrounds, roofs, walkways, and storage areas. The directly connected areas are indicated as “connected” or “draining to impervious” and do not include the pervious area or the impervious areas that drain to pervious areas. As expected, the land uses with the least impervious cover are open space (vacant land, cemeteries, golf courses) and low-density residential, and the land uses with the largest impervious covers are commercial areas, followed by industrial areas. For a typical high-density residential land use in this region (having 15 or more units per hectare), the major land cover was found to be landscaped areas, subdivided into front- and backyard categories, while 25 percent of this land-use area is covered by impervious surfaces broken down into three major subcategories: roofs, streets, and driveways. The subareas making up each land use show expected trends, with roofs and streets being the predominant directly connected impervious covers in residential areas, and parking and storage areas also being important in commercial and industrial areas.

continues next page
### BOX 3-3 Continued

<table>
<thead>
<tr>
<th>Land Use</th>
<th>Directly Connected Impervious Cover (%)</th>
<th>Disconnected Impervious Cover (%)</th>
<th>Pervious Cover (%)</th>
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<tbody>
<tr>
<td>High-Density Residential</td>
<td>14 (streets and roof)</td>
<td>10 (roofs)</td>
<td>76 (front and rear landscaping)</td>
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<tr>
<td>Medium-Density Residential (&lt;1960 to 1980)</td>
<td>11 (streets and roofs)</td>
<td>8 (roofs)</td>
<td>81 (front and rear landscaping)</td>
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<td>14 (streets and roofs)</td>
<td>5 (roofs)</td>
<td>80 (front and rear landscaping)</td>
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<td>Low-Density Residential</td>
<td>6 (streets)</td>
<td>4 (roofs)</td>
<td>89 (front and rear landscaping)</td>
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<td>Apartments</td>
<td>21 (streets and parking)</td>
<td>22 (roofs)</td>
<td>58 (front and rear landscaping)</td>
</tr>
<tr>
<td>Multiple Families</td>
<td>28 (roofs, parking, and streets)</td>
<td>7 (roofs)</td>
<td>65 (front and rear landscaping)</td>
</tr>
<tr>
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<td>59 (parking, streets, and roofs)</td>
<td>3 (parking)</td>
<td>39 (front and rear landscaping)</td>
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<td>Shopping Centers</td>
<td>64 (parking, roofs, and streets)</td>
<td>4 (roofs)</td>
<td>31 (front landscaping)</td>
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<tr>
<td>Schools</td>
<td>16 (roofs and parking)</td>
<td>20 (playground)</td>
<td>64 (front and rear landscaping, large turf)</td>
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<tr>
<td>Churches</td>
<td>53 (parking and streets)</td>
<td>7 (parking)</td>
<td>40 (front landscaping)</td>
</tr>
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<td>18 (storage and roofs)</td>
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<tr>
<td>Parks</td>
<td>32 (streets and parking)</td>
<td>33 (playground)</td>
<td>34 (large turf and undeveloped)</td>
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<td>2 (streets)</td>
<td>4 (roofs)</td>
<td>95 (large turf)</td>
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<tr>
<td>Vacant</td>
<td>5 (streets)</td>
<td>1 (driveways)</td>
<td>94 (undeveloped and large turf)</td>
</tr>
</tbody>
</table>

HYDROLOGIC AND GEOMORPHIC CHANGES

The watershed provides an organizing framework for the management of stormwater because it determines the natural patterns of water flow as well as the constituent sediment, nutrient, and pollutant loads. In undeveloped watersheds, hillslope hydrologic flow-path systems co-evolve with microclimate, soils, and vegetation to form topographic patterns within which ecosystems are spatially arranged and adjusted to the long-term patterns of water, energy, and nutrient availability. The landforms that comprise the watershed include the network patterns of streams, rivers, and their associated riparian zones and floodplains, as well as component freshwater lakes, reservoirs, wetlands, and estuaries.

This section starts with a discussion of precipitation measurement and characteristics before turning to the typical changes in hydrology and geomorphology of the watershed brought on by urbanization. In both the terrestrial and aquatic phases, retention and residence time of sediment and solutes decreases with increasing flow volume and velocity. This results in relatively high retention and low export of water and nutrients in undeveloped watersheds compared to decreasing retention and greater pollutant export in disturbed or developed systems.

The Storm in Stormwater

The magnitude and frequency of stormwater discharges are not just determined by rainfall. Instead, they are the combined product of storm and inter-storm characteristics, land use, the natural and built drainage system, and any stormwater control measures (SCMs) that have been implemented. The total volume and peak discharge of runoff, as well as the mobilization and transport of pollutants, are dependent on all aspects of the storm magnitude, catchment antecedent moisture conditions, and the interstorm period. Therefore, information on the frequency distribution of storm events and properties is an important aspect of understanding the distribution of pollutant concentrations and loads in stormwater discharges. In northern climates, runoff production from precipitation can be significantly delayed by the accumulation, ripening, and melt of snowpacks, such that much of the annual load of certain pollutants may be mobilized in peak flow from snowmelt events. Therefore, measurement of precipitation and potential accumulation in both liquid and solid form is critical for stormwater assessment.

Precipitation Measurements

Any given storm is characterized by the storm’s total rainfall (depth), its duration, and the average and peak intensity. A storm hyetograph depicts meas-
ured precipitation depth (or intensity) at a precipitation gauge as a function of time; an example is shown in Figure 3-5. This figure illustrates the typical high degree of variability of precipitation over the total duration of a storm. In this example, the total storm depth is 50.9 mm, the duration is 19 hours, and the peak intensity is 0.56 mm/minute (peak depth of 2.79 mm divided by the measurement increment of 5 minutes). The average intensity is 0.045 mm/minute, quite a bit lower than the peak intensity, since the storm duration is punctuated by periods of low and no measurable precipitation.

FIGURE 3-5  Example of a storm hyetograph at location RG2, September 20–21, 2001, Valley Creek watershed, Chester County, Pennsylvania. The time increment of measurement is 5 minutes, while the entire duration of this storm is about 16 hours.
In addition to measurements of individual storm events, precipitation data are routinely collected for longer time periods and compiled and analyzed annually when trying to understand local rainfall patterns and their impact on baseflow, water quality, and infrastructure design. Figure 3-6 shows the rainfall during 2007 at both humid (Baltimore) and arid (Phoenix) locations. Especially apparent in the Baltimore data is the fact that the majority of storm events are less than 20 mm in depth.

Several networks of precipitation gauges are available in the United States; gauge data are available online from the National Climatic Data Center (NCDC) (http://ncdc.nws.noaa.gov). High-resolution precipitation data (i.e., with measurement intervals of an hour or less) are typically not recorded except at primary weather service meteorological stations, while daily precipitation records are more extensively collected and available through the Cooperative Weather Observer Program (http://www.nws.noaa.gov/om/coop/). This distinction is important to stormwater managers because most stormwater applications require short-duration measurements or model results (minutes to hours). Fortunately, a combination of precipitation gauges and precipitation radar estimates are available to estimate precipitation depth and duration, as well as additional methods to estimate snowfall and snowpack water equivalent depth and conditions. (A thorough description of precipitation measurement by radar is given by Krajewski and Smith [2001]). While most of the conterminous United States is covered by NEXRAD radar for estimation of high-temporal-resolution precipitation at current resolutions of ~4 km, the radar backscatter information requires calibration and correction with precipitation gauge data, and satellite estimates

of precipitation are generally not sufficiently reliable for stormwater applications. It goes without saying that the measurement, quality assurance, and maintenance of long-term precipitation records are both vital and nontrivial to stormwater management.

Precipitation Statistics

The basic characterization of precipitation is by depth-duration-frequency curves, which describe the return period, recurrence interval, and exceedance probability (terms all denoting frequency) of different precipitation intensities (depths) over different durations. The methodology for determining the curves is described in Box 3-4. Precipitation durations of interest in stormwater management range from a few minutes (important for determining peak discharge from small urban drainage areas) to a year (where the interest is in the total annual volume of runoff production). As an example, one might be interested in the return period of the 1-inch, 1-hour event, or the 1-inch, 24-hour event; the latter would have a much shorter return period, because accumulating an inch of rain over a day is much more common than accumulating the same amount over just an hour.

**BOX 3-4**

Determining Depth-Duration-Frequency Curves

Depth-duration-frequency curves are developed from precipitation records using either annual maximum data series or annual exceedance data series. Annual maximum data series are calculated by extracting the annual maximum precipitation depths of a chosen duration from a record. In cases where there are only a few years of data available (less than 20 to 25 years), then an annual exceedance series (a type of “partial duration series”) for each storm duration can be calculated, where N largest values from N years are chosen. An annual maximum series excludes other extreme values of record that may occur in the same year. For example, the second highest value on record at an observing station may occur in the same year as the highest value on record but will not be included in the annual maximum series. The design precipitation depths determined from the annual exceedance series can be adjusted to match those derived from an annual maximum series using empirical factors (Chow et al., 1988; NOAA Atlas data series, see http://www.weather.gov/oh/hdsc/currentpf.htm, e.g., Bonnin et al., 2006). Hydrologic frequency analysis is then applied the data series to determine desired return periods by fitting a probability distribution to the data to determine the return periods\(^1\) of interest. The process is repeated for other chosen storm durations.

\(^1\)Analysis of annual maximum series produces estimates of the average period between years when a particular value is exceeded (“average recurrence interval”). Analysis of partial duration (annual exceedance) series gives the average period between cases of a particular magnitude (“annual exceedance probability”). The two results are numerically similar at rarer average recurrence intervals but differ at shorter average recurrence intervals (below about 20 years). NOAA (e.g., Bonnin et al., 2006) notes that the use of the terminology “average recurrence interval” and “annual exceedance probability” typically reflects the analysis of the two different series, but that sometimes the term “average recurrence interval” is used as a general term for ease of reference.
The National Weather Service has developed an online utility to estimate the return period for a range of depth-duration events for any place in the conterminous United States (http://hdsc.nws.noaa.gov/hdsc/pfds/). Figures 3-7 and 3-8 show examples of precipitation depth-duration-frequency curves for a humid location (Baltimore, Maryland) and an arid site (Phoenix, Arizona). As an illustration of the climatic influence on the depth-duration-frequency curves, the 2-year, 1-hour storm is associated with a depth of 1.2 inches of precipitation in Baltimore, whereas this same recurrence interval and duration are associated with a depth of only 0.6 inch of precipitation in Phoenix. Durations from 5 minutes to one day are shown because this is the range typically used in the design of stormwater management facilities. The shorter durations provide expected magnitude and frequency for brief but significant precipitation intensity peaks that can mobilize and transport large amounts of pollutants and erode soil, and they are used in high-resolution stormwater models. More commonly, however, stormwater regulations are written for 24-hour durations at 2-, 10-, 25-, 50-, or 100-year recurrence intervals.

Because storm magnitudes and frequencies vary by climatic region, it is reasonable to expect them to change during recurring climate events (e.g., El Niño) or over the long term by climate change. Alteration in convective precipitation by major urban centers has been documented for some time (Huff and Changnon, 1973). Some evidence exists that precipitation regimes are shifting systematically toward an increase in more intense rainfall events, which is consistent with modeled projections of global climate change increases in hydrologic extremes. Kunkel et al. (1999) analyzed precipitation data from 1,295 weather stations from 1931 to 1996 across the contiguous United States and found that storms with extreme levels of precipitation have increased in frequency. The analysis considered short-duration events (1, 3, and 7 days) of 1-year and 5-year return intervals. A linear trend analysis using Kendall’s slope estimator statistic indicated that the overall trend in 7-day, 1-yr events for the conterminous United States is upward at a rate of about 3 percent per decade for 1931 to 1996; the upward trend in 7-day, 5-year events is about 4 percent per decade. These two time series are shown in Figure 3-9. An increased frequency of intense precipitation events will shift depth-frequency-duration curves for a given location, with a given return period being associated with a more intense event. Alternatively, the return period for a given intensity (or depth) of an event will be reduced if the event is occurring more frequently. In light of climate change, depth-duration-frequency curves will need to be updated regularly in order to ensure that stormwater management facilities are not underdesigned for an increasing intensity of precipitation. Additional implications of climate change for stormwater management are discussed in Box 3-5.
FIGURE 3-7  Depth-duration-frequency curves for Baltimore, Maryland.  SOURCE: Data from the National Weather Service.

FIGURE 3-8  Depth-duration-frequency curves for Phoenix, Arizona.  SOURCE: Data from the National Weather Service.
FIGURE 3-9  Nationally averaged annual U.S. time series of the number of precipitation events of 7-day duration exceeding 1-year (dots) and 5-year (diamonds) recurrence intervals. SOURCE: Reprinted, with permission, from Kunkel et al. (1999). Copyright 1999 by American Meteorological Society.

BOX 3-5
Climate Change and Stormwater Management

An ongoing report series issued by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research summarizes the evidence for climate change to date and expected impacts of climate change, including impacts on the water resources sector (http://www.climatescience.gov/). According to the Intergovernmental Panel on Climate Change (IPCC 2007), annual precipitation will likely increase in the northeastern United States and will likely decrease in the southwestern United States over the next 100 years. In the western United States, precipitation increases are projected during the winter, whereas decreases are projected for the summer. As temperatures warm, precipitation will increasingly fall as rain rather than snow, and snow season length and snow depth are very likely to decrease in most of the country. More extreme precipitation events are also projected, which, when coupled with an anticipated increase in rain-on-snow events, would contribute to more severe flooding due to increases in extreme stormwater runoff.

The predictions for increases in the intensity and frequency of extreme events have significant implications for future stormwater management. First, many of the design standards currently in use will need to be revised, since they are based on historical data. For example, depth-duration-frequency curves used for design storm data will need to be updated, because the magnitude of the design storms will change. Even with revised design standards, in light of future uncertainty, new SCMs will need to be designed conservatively to allow for additional storage that will be required for regions with predicted trends in increased precipitation. In addition, existing SCM designs based on old standards may prove to be undersized in the future. Implementation of a monitoring program to check existing SCM inflows against original design inflows may be prudent to aid in judging whether retrofit of existing facilities or additional stormwater infrastructure is needed.
Design Storms

Given that only daily precipitation records are widely available, but short-duration data are required for stormwater analysis and prediction, *design storms* have been developed for the different regions of the United States by different state and federal resource agencies. A *design storm* is a specified temporal pattern of rainfall at a location, created using an overall storm duration and frequency relevant to the design problem at hand. Examples of design storms include the 24-hour, 100-year event for flood control and the 24-hour, 2-year event for channel protection. The magnitude of the design storm can be derived from data at a single gauge, or from synthesized regional data published by state or federal agencies. The simplest form of a design storm is a triangular hyetograph where the base is the duration and the height is adjusted so that the area under the curve equals the total precipitation. In instances where the hyetograph is to be used to estimate sequences of shorter duration intensities (i.e., minutes to a few hours) within larger duration events, depth-duration-frequency curve data can be used to synthesize a design storm hyetograph (see Chow et al., 1988). An example design storm for the 100-year storm event for St. Louis based on NOAA Atlas 14 depth-duration-frequency data is shown in Figure 3-10.

![Design Storm](image)

*FIGURE 3-10  Hundred-year design storm for St. Louis based on NOAA Atlas 14 data. SOURCE: Hoblit et al. (2004) based on data from Bonnin et al. (2003).*

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Conversion of Precipitation to Runoff

Dynamics of Watershed Flowpaths

Precipitation falling on the land surface is subject to evaporative loss to the atmosphere by vegetation canopy and leaf litter interception, evaporation directly from standing water on the surface and upper soil layers or impervious surfaces, and later transpiration through root uptake by vascular plants. Snowpack is also subject to sublimation (conversion of snow or ice directly to vapor), which results in the loss of a portion of the snow prior to melt. The rate of evaporative loss depends on local weather conditions (temperature, humidity, wind speed, solar radiation) and the rate and duration of precipitation. Precipitation (or snowmelt) in excess of interception and potential evaporative loss rates is then partitioned into infiltration and direct runoff.1

There is a gradation of flowpaths transporting water, sediment, and solutes through a watershed, ranging from rapid surface flowpaths through generally slower subsurface flowpaths. Residence times generally increase from surface to subsurface flowpaths, with rapid surface flow providing the major contribution to flood flow while subsurface flowpaths contribute to longer-term patterns of surface wetness. Watershed characteristics that influence the relative dominance of surface versus subsurface flowpaths include infiltration capacity as affected by land cover, soil properties, and macropores; subsurface structure or soil horizons with varying conductivity; antecedent soil moisture and groundwater levels; and the precipitation duration and intensity for a particular storm.

The distribution and activity of flowpaths result in changing patterns of soil moisture and groundwater depth, which result in patterns of soil properties, vegetation, and microbial communities. These ecosystem patterns, in turn, can have strong influences on the hydraulics of flow and biogeochemical transformations within the flowpaths, with important implications for sources, sinks, and transport of solutes and sediment in the watershed. Riparian areas, wetlands, and the benthos of streams and waterbodies are nodes of interaction between surface and groundwater flowpaths, yielding reactive environments in which “hot spots” of biogeochemical transformation develop (McClain et al., 2003). Thus, any alteration of surface and subsurface hydrologic flowpaths, for

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1 The term runoff is often used in two senses. For a given precipitation event, direct storm runoff refers to the rainfall (minus losses) that is shed by the landscape to a receiving waterbody. In an area of 100 percent imperviousness, the runoff nearly equals the rainfall (especially for larger storms). Over greater time and space scales, surface water runoff refers to streamflow passing through the outlet of a catchment, including base flow from groundwater that has entered the stream channel. The raw units of runoff in either case are volume per time, but the volumetric flowrate (discharge) is often divided by contributing area to express runoff in units of depth per time. In this way, unit runoff rates from various-sized watersheds can be compared to account for differences other than the contributing area.
example due to urbanization, not only alters the properties of soil and vegetation canopy but also reforms the ecosystem distribution of biogeochemical transformations.

Runoff Measurements

Surface water runoff for a given area is measured by dividing the discharge at a given point in the stream channel by the contributing watershed area. The basic variables describing channel hydraulics include width, mean depth, slope, roughness, and velocity. Channel discharge is the product of width, depth, and velocity and is typically estimated by either directly measuring each of these three components, or by development of a rating curve of measured discharge as a function of water depth, or stage relative to a datum, of the channel that is more easily estimated by a staff gauge or pressure transducer. The establishment of a gauging station to measure discharge typically requires a stable cross section so that stage can be uniquely related to discharge. Maintenance of reliable, long-term gauge sites is expensive and requires periodic remeasurement to update rating curves, as well as to remove temporary obstructions that may raise stage relative to unobstructed conditions.

Most stream gauging in the United States is carried out by the USGS, and can be found on-line at [http://waterdata.usgs.gov/nwis](http://waterdata.usgs.gov/nwis). Recent reviews of standard methods of stream gauging and the status of the USGS stream gauging network are given by the USGS (1998) and the National Research Council (NRC, 2004). A major concern is the overall decline in the number of active gauges, particularly long-term gauges, as well as the representativeness of the stream gauge network relative to the needs of stormwater permitting. For example, restored streams typically lack any gauged streamflow or water quality information prior to or following restoration. This makes it very difficult to assess both the potential for successful restoration and whether project goals are met.

Support of existing and development of new gauges is often in collaboration through a co-funding mechanism with other agencies. Municipal co-funding for stations in support of National Pollutant Discharge Elimination System (NPDES) permitting is common and has tended to shift the concentration of active gauges toward more urban areas. Note that the USGS river monitoring system was originally designed for resource inventory, and therefore did not originally sample many headwater streams, particularly intermittent and ephemeral channels that are typically most proximal to stormwater discharges. While this is beginning to change with municipal co-funding, headwater streams are still underrepresented in the National Water Information System relative to their ecological significance.

Reliable records for stream discharge are vital because the frequency distribution and temporal trends of flows must be known to evaluate long-term loading to waterbodies. Magnitude and frequency analysis of sediment and other
stream constituent loads consists of a transport equation as a function of discharge, integrated over the discharge frequency distribution (e.g., Wolman and Miller, 1960). Different constituent loads have different forms of dependency on discharge, but are often nonlinear such that long-term or expected loads cannot be simply evaluated from mean flow conditions. Similar to precipitation, discharge levels often follow an Extreme Value distribution, dependent on climate, land use, and hydrogeology, but which is typically dampened compared to precipitation due to the memory effects of subsurface storage and flows (e.g., Winter, 2007).

**Impacts of Urbanization on Runoff**

*Shift from Infiltration and Evapotranspiration to Surface Runoff*

Replacement of vegetation with impervious or hardened surfaces affects the hydrologic budget—the quantity of water moving through each component of the hydrologic cycle—in a number of predictable ways. As the percent of the landscape that is paved over or compacted is increased, the land area available for infiltration of precipitation is reduced, and the amount of stormwater available for direct surface runoff becomes greater, leading to increased frequency and severity of flooding. Reduced infiltration of precipitation leads to reduced recharge of the groundwater reservoir; absent new sources of recharge, this can lead to reduction in base flow of streams (e.g., Simmons and Reynolds, 1982; Rose and Peters, 2001). Vegetation removal also results in a lower amount of evapotranspiration compared to undeveloped land. This can have particularly profound hydrologic effects in those regions of the country where a significant percent of precipitation is evapotranspired, such as the arid Southwest (Ng and Miller, 1980). Figure 3-11 illustrates the changes to these components of the hydrologic budget as the percent of impervious area is increased.

It should be noted that the conversion in hydrology from infiltrated water to surface runoff following urbanization is not entirely straightforward in all cases. Leaking pressurized water supply pipes and sanitary sewers, subsurface discharge of septic system effluent (Burns et al., 2005), infiltration of stormwater from unlined detention ponds, and lawn irrigation can offset reduced infiltration of precipitation, such that stream baseflow levels may actually be increased, especially during low base flow months, when such effects would be most pronounced (Konrad and Booth, 2005; Meyer, 2005). Cracks in sealed surfaces can also provide concentrated points of infiltration (Sharp et al., 2006).
FIGURE 3-11  As land cover changes from vegetated and undeveloped (upper left) to developed with increased connected impervious surfaces (lower right), the partitioning of precipitation into other components of the hydrologic cycle is shifted. Evapotranspiration and shallow and deep infiltration are reduced, and surface runoff is increased. SOURCE: Adapted from the Federal Interagency Stream Restoration Working Group (FISRWG, 2000).

Relationship Between Imperviousness, Drainage Density, and Runoff

Excess runoff due to urbanization is a direct reflection of the land uses onto which the precipitation falls, as well as the presence of drainage systems that receive stormwater from many separate source areas before it enters receiving waters. Thus, a functional way of partitioning urban areas is by the nature of the impervious cover and by its connection to the drainage system, underlying the differentiation of total impervious area and effective impervious area discussed in Box 1-2.

As examples of how runoff changes with urbanization, Figure 3-12 shows
daily stream flow values for a low-density suburban catchment and a high-
density urban catchment in the Baltimore, Maryland area. The low-density site
(Figure 3-12A) shows a strong seasonal signal and a marked decline in flow
during an extreme drought in 2002. In contrast, the more densely urbanized
catchment (Figure 3-12B) shows a much greater variability in flow that is domi-
nated by impervious surface runoff, and a dampened response to the drought
because natural groundwater flow is a much smaller component of the total dis-
charge.

The percentage of time a discharge level is equaled or exceeded is displayed
by flow duration curves, which show the cumulative frequency distributions of
flows for a given duration. Examples for three catchments in the Baltimore area
are given in Figure 3-13, showing the tendency for urban areas to produce high
flows with much longer aggregate durations.

As another example of how runoff changes with imperviousness, a locally
calibrated version of WinSLAMM was used to investigate the relationships be-
tween watershed and runoff characteristics for 125 individual neighborhoods in

FIGURE 3-12  Daily time series of flows in (A) a low-density suburban and forested catch-
ment (Baisman Run, http://waterdata.usgs.gov/md/nwis/uv/?site_no=01583580) and (B) a
catchment dominated by medium- to high-density residential and commercial land uses
(Dead Run, http://waterdata.usgs.gov/md/nwis/uv/?site_no=01589330). Both lie within the
Piedmont physiographic province.
Jefferson County, Alabama (Bochis-Micu and Pitt, 2005). Figure 3-14 shows the relationships between the directly connected impervious area values and the calculated volumetric runoff coefficient ($R_v$, which is the volumetric fraction of the rainfall that occurs as runoff), based on 43 years of local rain data. As expected, there is a strong relationship between these parameters for both sandy and clayey soil conditions. It is interesting to note that the $R_v$ values are relatively constant until values of directly connected impervious cover of 10 to 15 percent are reached (at $R_v$ values of about 0.07 for sandy soil areas and 0.16 for clayey soil areas)—the point where receiving water degradation typically has been observed to start (as discussed later in the chapter). The 25 to 30 percent directly connected impervious levels (where significant degradation is usually observed) is associated with $R_v$ values of about 0.14 for sandy soil areas and 0.25 for clayey soil areas; this is where the curves start to greatly increase in slope.

**Relationship Between Runoff and Rainfall Conditions**

The runoff that results from various land uses also varies depending on rainfall conditions. For small rain depths, almost all the runoff originates solely from directly connected impervious areas, as disconnected areas have most of their flows infiltrated (Pitt, 1987). For larger storms, both directly connected
FIGURE 3-14 Relationships between the directly connected impervious area (%) and the calculated volumetric runoff coefficients (R_v) for sandy soil (top) and clayey soil (bottom). SOURCE: Reprinted, with permission, from Bochis-Micu and Pitt (2005). Copyright 2005 by Water Environment Federation, Alexandria, Virginia.
and disconnected impervious areas contribute runoff to the stormwater management system. For example, Figure 3-15 (created using WinSLAMM; Pitt and Voorhees, 1995) shows the relative runoff contributions for a large commercial/mall area in Hoover, Alabama, for different rains (Bochis, 2007). In this example, about 80 percent of the runoff originates from the parking areas for the smallest runoff-producing rains. This contribution decreases to about 55 percent at rain depths of about 0.5 inch (13 mm). This decrease in the importance of parking areas as a source of runoff volume is associated with an increase in runoff contributions from streets and directly connected roofs. In many areas, pervious areas are not hydrologically active until the rain depths are relatively large and are not significant runoff contributors until the rainfall exceeds about 25 mm for many land uses and soil conditions. However, compacted urban soils can greatly increase the flow contributions from pervious areas during smaller rains. Burges and others (1998), for example, found that more than 60 percent of the storm runoff in a suburban development in western Washington State originated from nominally “green” parts of the landscape, primarily lawns.

A further example illustrating the relationship between rainfall and runoff is given for Milwaukee, summarized in Box 3-6. The two curves of Figure 3-16 show a relationship between rainfall and runoff that is typical of urban areas. Very small storms (< 0.05 inch) produce no measurable runoff, owing to removal by interception storage and evaporation. Storms that deposit up to one inch of rainfall constitute about 90 percent of the storm events in this region, but these events produced only about 50 percent of the runoff. Very large events (greater than 3 inches of precipitation) are rare and destructive, accounting for only a few percent of the annual rainfall events.

![Figure 3-15](image-url)  
**FIGURE 3-15** Surfaces contributing to runoff for a commercial/mall area. **SOURCE:** Reprinted, with permission, from Bochis (2007). Copyright 2007 by Celina Bochis.
Example Rainfall and Runoff Distributions

Figure 3-16 is an example of rainfall and runoff observed at Milwaukee, Wisconsin (Bannerman et al., 1983), as monitored during the Nationwide Urban Runoff Program (NURP) (EPA, 1983). This observed distribution is interesting because of the unusually large rains that occurred twice during the monitoring program. These two major rains would be in the category of design storms for conventional drainage systems. These plots indicate that these very large events, in the year they occurred, caused a measurable fraction of the annual pollutant loads and runoff volume discharges, but smaller events were responsible for the vast majority of the discharges. In typical years, when these rare design events do not occur, their pro-rated contributions would be even smaller.

More than half of the runoff from this typical medium-density residential area was associated with rain events that were smaller than 0.75 inch. Two large storms (about 3 and 5 inches in depth), which are included in the figure, distort this figure because, on average, the Milwaukee area only expects one 3.5-inch storm about every five years, and 5-inch storms even less frequently. If these large rains did not occur, such as for most years, then the significance of the smaller rains would be even greater. The figure also shows the accumulated mass discharges of different pollutants (suspended solids, chemical oxygen demand [COD], phosphates, and lead) monitored during the Milwaukee NURP project. When these figures are compared, it is seen that the runoff and pollutant mass discharge distributions are very similar and that variations in the runoff volume are much more important than variations in pollutant concentrations (the mass divided by the runoff volume) for determining pollutant mass discharges.

These rainfall and runoff distributions for Milwaukee can thus be divided into four regions:

- **Less than 0.5 inch.** These rains account for most of the events, but little of the runoff volume, and they are therefore easiest to control. They produce much less pollutant mass discharge and probably have less receiving water effects than other rains. However, the runoff pollutant concentrations likely exceed regulatory standards for several categories of critical pollutants (bacteria and some total recoverable heavy metals). They also cause large numbers of overflow events in uncontrolled combined sewers. These rains are very common, occurring once or twice a week (accounting for about 60 percent of the total rainfall events and about 45 percent of the total runoff-generating events), but they only account for about 20 percent of the pollutant mass discharges.
annual runoff and pollutant discharges. Rains less than about 0.05 inch did not produce noticeable runoff.

- **0.5 to 1.5 inches.** These rains account for the majority of the runoff volume (about 50 percent of the annual volume for this Milwaukee example) and produce moderate to high flows. They account for about 35 percent of the annual rain events, and about 20 percent of the annual runoff events, by number. These rains occur on average about every two weeks from spring to fall and subject the receiving waters to frequent high pollutant loads and moderate to high flows.

- **1.5 to 3 inches.** These rains produce the most damaging flows from a habitat destruction standpoint and occur every several months (at least once or twice a year). These recurring high flows, which were historically associated with much less frequent rains, establish the energy gradient of the stream and cause unstable streambanks. Only about 2 percent of the rains are in this category, but they are responsible for about 10 percent of the annual runoff and pollutant discharges.

- **Greater than 3 inches.** The rains in this category are included in design storms used for traditional drainage systems in Milwaukee, depending on the times of concentration and rain intensities. These rains occur only rarely (once every several years to once every several decades, or less frequently) and produce extremely large flows that greatly exceed the capacities of the storm drainage systems, causing extensive flooding. The monitoring period during the Milwaukee NURP was unusual in that two of these events occurred. Less than 2 percent of the rains were in this category (typically <<1 percent would be in this category), and they produced about 15 percent of the annual runoff quantity and pollutant discharges. However, when they do occur, substantial property and receiving water damage results (mostly associated with habitat destruction, sediment scouring, and the flushing of organisms great distances downstream and out of the system). The receiving water can conceivably recover naturally to pre-storm conditions within a few years. These storms, while very destructive, are sufficiently rare that the resulting environmental problems do not justify the massive controls that would be necessary to decrease their environmental effects.

**Alteration of the Drainage Network**

As shown in Figure 3-17, urbanization disrupts natural systems in ways that further complicate the hydrologic budget, beyond the imperviousness effects on runoff discussed earlier. As an area is urbanized, lower-order stream channels are typically re-routed or encased in pipes and paved over, resulting in a highly altered drainage pattern. The buried stream system is augmented by an extensive system of storm drains and pipes, providing enhanced drainage density (total lengths of pipes and channels divided by drainage area) compared to the natural system. Figure 3-18 shows how the drainage density of Baltimore today compares to the natural watershed before the modern stormwater system was fully developed. The artificial drainage system occupies a greater percentage of the landscape compared to natural conditions, permanently altering the terrestrial component of the hydrologic cycle.
Flowpaths are altered in other ways by urban infrastructure. Buried stormwater and sewer pipes can act as infiltration galleries for groundwater, causing shortened groundwater flowpaths between groundwater reservoirs and stream systems. Natural surface water pathways are often interrupted or reversed, as shown by the blue lines in Figure 3-19 for a drainage system in Baltimore. Understanding how the system operates as a whole can often require knowledge of the history of construction conditions and field verification of the actual flow paths.

Large-scale infrastructure such as dams, ponds, and bridges can also have a major impact on stormwater flows. Figure 3-20 illustrates the interruption of the drainage network by bridges and culverts, even in places where there have been attempts to keep excessive development out of the riparian corridor. Simulations and post-flood mapping in areas around Baltimore have shown that bridge abutments such as those shown in Figure 3-20 can slow down channel floodwaters during storms. This is because water backs up behind bridges constructed...
FIGURE 3-18 Baltimore City before and after development of its stormwater system. The left-hand panel shows first- and second-order streams lost to development. The right-hand panel shows the increase in drainage density resulting from construction of the modern storm-drain network. SOURCE: Courtesy of William Stack, Baltimore Department of Public Works.
FIGURE 3-19 Dead Run drainage system, Baltimore, Maryland. Black lines indicate surface (daylighted) drainage; dark grey indicates the subsurface storm-drain system. The surface drainage system is highly disconnected. From the coverage it is difficult to impossible to discern the flow direction of some of the surface drainage components. SOURCE: Reprinted, with permission, from Meierdierks et al. (2004). Copyright 2004 by the American Geophysical Union.
across the floodplain and spreads out over land surfaces and then flows back into channels as floodwaters subside. Although reducing the severity of downstream flooding, this phenomenon also interrupts the transport of sediment, leading to local zones of both enhanced deposition and downstream scour.

**Alteration of Travel Times**

The combination of impervious surface and altered drainage density provides significantly more rapid hydraulic pathways for stormwater to enter the nearest receiving waterbody compared to a natural landscape. This is illustrated quantitatively by Figure 3-21, which shows that the lag time—the difference in time between the center of mass of precipitation and the center of mass of the storm response hydrograph—is reduced for an urbanized landscape compared to a natural one.

The increase in surface runoff volumes and reduction in lag times between
precipitation and a waterbody’s response give rise to greater velocities and volumetric discharges in receiving waters. Storm hydrographs in a developed setting peak earlier and higher than they do in undeveloped landscapes. This altered flow regime is of concern to property owners because upstream development can increase the probability of a flood-prone property being inundated. Properties in the floodplain and near stream channels are particularly susceptible to flooding from upstream development. Such increased flood risk is accompanied by associated potential property damages and costs of replacement or repair.

Various descriptors can be used to quantify the effects of urbanization on streamflow including flood frequency, flow duration, mean annual flood, discharge at bankfull stage, and frequency of bankfull stage. The “classic” view of
urban-induced changes to runoff was presented by Leopold (1968), who provided several quantitative descriptors of the effects of urbanization on the mean annual flood. For example, Figure 3-22 shows the ratio of discharge before and after urbanization for the mean annual flood for a 1-square-mile area as a function of percentage of impervious area and percentage area served by a storm-drain system. This shows that for unserved areas, increases from 0 to 100 percent impervious area will increase the peak discharge by a factor of 2.5. However, for 100 percent sewer areas, the ratio of peak discharges ranges from 1.7 to 8 for 0 to 100 percent impervious area. Clearly both impervious surfaces and the presence of a storm-drain system combine to increase discharge rates in receiving waters. Combining this information with regional flood frequency data, a discharge–frequency relationship can be developed that shows the expected discharge and recurrence interval for varying degrees of storm-drain coverage and impervious area coverage. An example is shown in Figure 3-23, using data from the Brandywine Creek watershed in Pennsylvania (Leopold, 1968). Bank-full flow for undeveloped conditions in general has a recurrence interval of about 1.5 years (which, in the particular case of the Brandywine, was 67 cubic feet per second); with 40 percent of the watershed area paved, this discharge would occur about three times as often.

FIGURE 3-22  Ratio of peak discharge after urbanization to peak discharge before urbanization for the mean annual flood for a 1-square-mile drainage area, as a function of percent impervious surface and percent area drained by storm sewers. SOURCE: Leopold (1968).
Over the past four decades since this first quantitative characterization of urban hydrology, a much greater variety of hydrologic changes resulting from urbanization has been recognized. Increases in peak discharge are certainly among those changes, and they will always gather attention because of their direct impact on human infrastructure and potential for more frequent and more severe flooding. The extended duration of flood flows, however, also affects natural channels because of the potential increase in erosion. Ecological effects of urban-altered flow regimes are even more diverse, because changes in the sequence and frequency of high flows, the rate of rise and fall of the hydrograph, and even the season of the year in which high flows can occur all have significant ecological effects and can be dramatically altered by watershed urbanization (e.g., Rose and Peters, 2001; Konrad et al., 2005; Roy et al., 2005; Poff et al., 2006).
The overarching conclusion of many studies is that the impact of urbanization on the hydrologic cycle is dramatic. Increased impervious area and drainage connectedness decreases stormwater travel times, increases flow rates and volumes, and increases the erosive potential of streams. The flooding caused by increased flows can be life-threatening and damaging to property. As described below, changes to the hydrologic flow regime also can have deleterious effects on the geomorphic form of stream channels and the stability of aquatic ecosystems. Although these impacts are commonly ignored in efforts to improve “water quality,” they are inextricably linked to measured changes in water chemistry and must be part of any attempt to recover beneficial uses that have been lost to upstream urbanization.

**Geomorphology**

Watershed geomorphology is determined by the arrangement, interactions, and characteristics of component landforms, which include the stream-channel network, the interlocking network of ridges and drainage divides, and the set of hillslopes between the channel (or floodplain) and ridge. The stream and ridge systems define complementary networks, with the ridge (or drainage divide) network separating the drainage areas contributing to each reach in the stream network. At the hillslope scale, the ridges provide upper boundaries of all surface flowpaths which converge into the complementary stream reaches. A rich literature describes the topology and geometry of stream and ridge networks (e.g., Horton, 1945; Strahler, 1957, 1964; Shreve, 1966, 1967, 1969; Smart, 1968; Abrahams, 1984; Rodriguez-Iturbe et al., 1992).

Besides stream channels, a variety of other water features and landforms make up a watershed. Fresh waterbodies (ponds, lakes, and reservoirs) are typically embedded within the stream network, while wetlands may be either embedded within the stream network or separated and upslope from the channels. Estuaries represent the interface of the stream network with the open ocean. Additional fluvial and colluvial landforms include alluvial fans, landslide features, and a set of smaller features within or near the channels and floodplains including bar deposits, levees, and terraces. Each of these landforms are developed and maintained by the fluvial and gravitational transport and deposition of sediment, and are therefore potentially sensitive to disruption or alteration of flowpaths, hydrologic flow regimes, and sediment supply.

**Stream Network Form and Ordering Methods**

Most watersheds are fully convergent, with tributary streams combining to form progressively larger channels downstream. The manner in which streams from different source areas join to produce mainstreams strongly influences the propagation of stormwater discharge and pollutant concentrations, and the con-
EFFECTS OF URBANIZATION ON WATERSHEDS

sequent level of ecological impairment in the aquatic ecosystem.

Methods for indexing the topologic position of individual reaches within the drainage network have been introduced by Horton (1945), Strahler (1957), Shreve (1966, 1967) and others. All stream topologic systems are dependent on the identification of first-order streams—the most upstream element of the network—and their lengths and drainage areas. Unfortunately, no universal standards exist to define where the stream head is located, or whether perennial, intermittent, and ephemeral channels should be considered in this determination. While this may seem like a trivial process, the identification and delineation of these sources effectively determines what lengths and sections of channels are defined to be waterbodies and, thus, the classification of all downstream waterbodies.

Nadeau and Rains (2007) have recently reviewed stream-channel delineation in the United States using standardized maps and hydrographic datasets to better relate climate to the extent of perennial, intermittent, and ephemeral channel types. Because this may influence the set of stream channels that are regulated by the Clean Water Act (CWA), it is the subject of current legal arguments in courts up to and including the Supreme Court (e.g., Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers, 531 U.S. 159 [2001], John A. Rapanos et al. vs. United States [U.S., No. 04-1034, 2005]). In addition to the stream-channel network, additional features (discussed below) that are embedded in or isolated from the delineated stream network (lakes, ponds, and wetlands) are subject to regulation under the CWA based on their proximity or interaction with the defined stream and river network. Therefore, definition of the extent and degree of connectivity of the nation’s stream network, with an emphasis on the headwater region, is a critical determinant of the set of waterbodies that are regulated for stormwater permitting (Nadeau and Rains, 2007).

Stream Reach Geomorphology

Within the channel network, stream reaches typically follow a regular pattern of changes in downstream channel form. Hydraulic geometry equations, first introduced by Leopold and Maddock (1953), describe the gross geomorphic adjustment of the channel (in terms of average channel depth and width) to the flow regime and sometimes the sediment supply. Within this general pattern of larger flows producing larger channels, variations in channel form are evident, particularly the continuum among straight, meandering, or braided patterns. These forms are dependent on the spatial and temporal patterns of discharge, sediment supply, transport capacity, and roughness elements.

Most natural channels have high width-to-depth ratios and complexity of channel form compared with engineered channels. Meanders are ubiquitous self-forming features in channels, created as accelerated flow around the outside of the meander entrains and transports more sediment, producing greater flow depths and eroding the bank, while decelerated flow on the inside of the mean-
der results in deposition and the formation of lower water depth and bank gradients. These channels typically show small-scale alternation between larger cross sections with lower velocities and defining pools, and smaller cross sections with higher velocity flow in riffles. Braided streams form repeated subdivision and reconvergence of the channel in multiple threads, with reduced specific discharge compared to a single channel. Natural obstructions including woody debris, boulders, and other large (relative to channel dimensions) features all contribute to hydraulic and habitat heterogeneity. The complexity of these channel patterns contributes to hydraulic roughness, further dissipating stream energy by increasing the effective wetted perimeter of the channel through a valley and deflecting flow between banks.

Embedded Standing Waterbodies

Standing waterbodies include natural, constructed, or modified ponds and lakes and are characterized by low or near-zero lateral velocity. They can be thought of as extensions of pools within the drainage network, although there is no clear threshold at which a pool can be defined as a pond or lake. When they are embedded within the channel network, they are characterized with much greater cross-sectional area (width x depth), lower surface water slopes (approaching flat), and lower velocities than a stream reach of similar length. Therefore, standing waterbodies function as depositional zones, have higher residence times, and provide significant storage of water, sediment, nutrients, and other pollutants within the stream network.

Riparian Zone

The riparian area is a transitional zone between the active channel and the uplands, and between surface water and groundwater. The area typically has shallower groundwater levels and higher soil moisture than the surrounding uplands, and it may support wetlands or other vegetation communities that require higher soil moisture. Riparian zones provide important ecosystem functions and services, such as reducing peak flood flows, transforming bioavailable nutrients into organic matter, and providing critical habitat.

In humid landscapes, a functioning riparian area commonly is an area where shallow groundwater forms discharge seeps, either directly to the surface and then to the stream channel or through subsurface flowpaths to the stream channel. The potential for high moisture and organic material content provides an environment conducive to anaerobic microbial activity, which can provide effective sinks for inorganic nitrogen by denitrification, reducing nitrate loading to the stream channel. However, the width of the effective riparian zone depends on local topographic gradients, hydrogeology, and the channel geomorphology (Lowrance et al., 1997). In steeply incised channels and valleys, or areas with
deeper flowpaths, the riparian zone may be narrow and relatively well drained.

Under more arid conditions with lower groundwater levels, riparian areas may be the only areas within the watershed with sufficient moisture levels to support significant vegetation canopy cover, even though saturation conditions may occur only infrequently. Subsurface flowpaths may be oriented most commonly from the channel to the bed and banks, forming the major source of recharge to this zone from periodic flooding. In monsoonal climates in the U.S. southwest, runoff generated in mountainous areas or from storm activity may recharge riparian aquifers well downstream from the storm or snowmelt activity. Channelization that reduces this channel-to-riparian recharge may significantly impair riparian and floodplain ecosystems that provide critical habitat and other ecosystem services (NRC, 2002).

Floodplains

The presence and distribution of alluvial depositional zones, including floodplains, is dependent on the distribution and balance of upstream sediment sources and sediment transport capacity, the temporal and spatial variability of discharge, and any geological structural controls on valley gradient. Lateral migration of streams contributes to the development of floodplains as the outer bank of the migrating channel erodes sediment and deposition occurs on the opposite bank. This leads to channels that are closely coupled to their floodplains, with frequent overbank flow and deposition, backwater deposits, wetlands, abandoned channels, and other floodplain features. During major events, overbank flooding and deposition adds sediment, nutrients, and contaminants to the floodplain surface, and may significantly rework preexisting deposits and drainage patterns. Constructional landforms typical of urbanized watersheds, such as levees, tend to disconnect streams from their floodplains.

Changes in Geomorphology from Urbanization

Changes to channel morphology are among the most common and readily visible effects of urban development on natural stream systems (Booth and Henshaw, 2001). The actions of deforestation, channelization, and paving of the uplands can produce tremendous changes in the delivery of water and sediment into the channel network. In channel reaches that are alluvial, the responses are commonly rapid and often dramatic. Channels widen and deepen, and in some cases may incise many meters below the original level of their beds. Alternatively, channels may fill with sediment derived from farther upstream to produce a braided form where a single-thread channel previously existed.

The clearest single determinant of urban channel change is the alteration of the hydrologic response of an urban watershed, notably the increase in streamflow discharges. Increases in runoff mobilize sediment both on the land surface
and within the stream channel. Because transport capacity increases nonlinearity
with flow velocity (Vogel et al., 2003), much greater transport will occur in
higher flow events. However, the low frequency of these events may result in
decreasing cumulative sediment transport during the highest flows, as described
by standard magnitude and frequency analysis (Wolman and Miller, 1960), such
that the maximum time-integrated sediment transport occurs at moderate flows
(e.g., bankfull stage in streams in the eastern United States).

If the increase in sediment transport caused by the shift in the runoff regime
is not matched by the sediment supply, channel bed entrenchment and bank ero-
sion and collapse lead to a deeper, wider channel form. Increases in channel
dimensions caused by increased discharges have been observed in numerous
studies, including Hammer (1972), Hollis and Luckett (1976), Morisawa and
LaFlure (1982), Neller (1988), Whitlow and Gregory (1989), Moscrip and
Montgomery (1997), and Booth and Jackson (1997). MacRae (1997), reporting
on other studies, found that channel cross-sectional areas began to enlarge after
about 20 to 25 percent of the watershed was developed, commonly correspond-
ing to about 5 percent impervious cover. When the watersheds were completely
developed, the channel enlargements were about 5 to 7 times the original cross-
sectional areas. Channel widening can occur for several decades before a new
equilibrium is established between the new cross-section and the new dis-
charges.

Construction results in a large—but normally temporary—increase in sedi-
ment load to aquatic systems (e.g., Wolman and Schick, 1967). Indeed, erosion
and sediment transport rates can reach up to more than 200 Mg/ha/yr on con-
struction sites, which is well in excess of typical rates from agricultural land
(e.g., Wolman and Schick, 1967; Dunne and Leopold, 1978); rates from undis-
turbed and well-vegetated catchments are negligible (e.g., <<1 Mg/ha/yr). The
increased sediment loads from construction exert an opposing tendency to chan-
nel erosion and probably explain much of the channel narrowing or shallowing
that is sometimes reported (e.g., Leopold, 1973; Nanson and Young, 1981; Ebi-

Additional sediment is commonly introduced into the channel network by
the erosion of the streambank and bed itself. Indeed, this source can become the
largest single fraction of the sediment load in an urbanizing watershed (Trimble,
1997). For example, Nelson and Booth (2002) reported on sediment sources in
the Issaquah Creek watershed, an urbanizing, mixed-use watershed in the Pacific
Northwest. Human activity in the watershed, particularly urban development,
has caused an increase of nearly 50 percent in the annual sediment yield, now
estimated to be 44 tons/km²/yr¹. The main sources of sediment in the watershed
are landslides (50 percent), channel-bank erosion (20 percent), and stormwater
discharges (15 percent).

The higher flow volumes and peak discharge caused by urbanization also
tend to preferentially remove fine-grained sediment, leaving a lag of coarser bed
material (armoring) or removing alluvial material entirely and eroding into the
geologic substrate (Figure 3-24). The geomorphic outcome of these changes is a
EFFECTS OF URBANIZATION ON WATERSHEDS

FIGURE 3-24 Example of an urban stream that has eroded entirely through its alluvium to expose the underlying consolidated geologic stratum below (Thornton Creek, Seattle, Washington). SOURCE: Derek Booth, Stillwater Sciences, Inc.

mix of erosional enlargement of some stream reaches, significant sedimentation in others, and potential head-ward downcutting of tributaries as discharge levels from small catchments increase. The collective effects of these processes have been described by Walsh et al. (2005) as “Urban Stream Syndrome,” which includes not only the visible alteration of the physical form of the channel but also the consequent deterioration of stream biogeochemical function and aquatic trophic structures.

Other changes also accompany these geomorphic changes. Episodic inundation of the floodplain during floods may be reduced in magnitude and frequency, depending on the increases in peak flow relative to the deepening and resultant increase in flow capacity of the channel. Where deeply entrenched, this channel morphology will lower the groundwater level adjacent to the channel. The effectiveness of riparian areas in filtering or removing solutes is thus reduced because subsurface water may reach the channel only by flowpaths now well below the organic-rich upper soil horizons. Removal of fine-grained stream-bottom sediment, or erosion down to bedrock, may substantially lower the exchange of stream water with the surrounding groundwater of the hyporheic zone.

In addition to these indirect effects on the physical form of the stream channel, urbanization also commonly modifies streams directly to improve drainage, applying channel straightening and lining to reduce friction, increase flow capacity, and stabilize channel position (Figure 3-25). The enlarged and often
lined and straightened stream-channel cross section reduces the complexity of the bed and the contact between the stream and floodplain, and increases transport efficiency of sediment and solutes to receiving waterbodies. Enhanced sedimentation of receiving waterbodies, in turn, reduces water clarity, decreases depth, and buries the benthic environment.

**POLLUTANT LOADING IN STORMWATER**

Hydrologic flowpaths influence the production of particulate and dissolved substances on the land surface during storms, as well as their delivery to the stream-channel network. Natural watersheds typically develop a sequence of ecosystem types along hydrologic flowpaths that utilize available limiting resources, thereby reducing their export farther downslope or downstream, such that in-stream concentrations of these nutrients are low. As a watershed shifts from having mostly natural pervious surfaces to having heavily disturbed soils, new impervious surfaces, and activities characteristic of urbanization, the runoff quality shifts from relatively lower to higher concentrations of pollutants. Anthropogenic activities that can increase runoff pollutant concentrations in urban watersheds include application of chemicals for fertilization and pest control; leaching and corrosion of pollutants from exposed materials; exhaust emissions,
leaks from, and wear of vehicles; atmospheric deposition of pollutants; and inappropriate discharges of wastes.

Most lands in the United States that have been developed were originally grasslands, prairies, or forest. About 40 percent of today’s developed land went through an agricultural phase (cropland or pastureland) before becoming urbanized, while more than half of today’s developed land area has been a direct conversion of natural covers (USDA, 2000). Agricultural land can produce stormwater runoff with high pollutant concentrations via soil erosion, the introduction of chemicals (fertilizers, pesticides, and herbicides), animal operations that are major sources of bacteria in runoff, and forestry operations. Indeed, urban stormwater may actually have slightly lower pollutant concentrations than other nonpoint sources of pollution, especially for sediment and nutrients. The key difference is that urban watersheds produce a much larger annual volume of runoff waters, such that the mass of pollutants discharged is often greater following urbanization. Some of the complex land-use–pollutant loading relationships are evident in Box 3-7, which shows the measured annual mass loads of nitrogen and phosphorus in four small watersheds of different land use monitored as part of the Baltimore Long-Term Ecological Research program. Depending on the nutrient and the year, the agricultural and urban watersheds had a higher nutrient export rate than the forested subwatershed.

Table 3-3 summarizes the comparative importance of urban land-use types in generating pollutants of concerns that can impact receiving waters (Burton and Pitt, 2002). This summary is highly qualitative and may vary depending on the site-specific conditions, regional climate, activities being conducted in each land use, and development characteristics. It should be noted that the rankings in Table 3-3 are relative to one another and classified on a per-unit-area basis. Furthermore, this table shows the parameters for each land-use category, such that the effects for a community at large would be dependent on the areas of each land use shown. Thus, although residential land use is shown to be a relatively smaller source of many pollutants, it is the largest fraction of land use in most communities, typically making it the largest stormwater source on a mass pollutant discharge basis. Similarly, freeway, industrial, and commercial areas can be very significant sources of many stormwater problems, and their discharge significance is usually much greater than their land area indicates. Construction sites are usually the overwhelming source of sediment in urban areas, even though they make up very small areas of most communities. A later table (Table 3-4) presents observed stormwater discharge concentrations for selected constituents for different land uses.

The following section describes stormwater characteristics associated with urbanized conditions. At any given time, parts of an urban area will be under construction, which is the source of large sediment losses, flow path disruptions, increased runoff quantities, and some chemical contamination. Depending on the time frame of development, increased stormwater pollutant discharges associated with construction activities may last for several years until land covers are stabilized. After construction has been completed, the characteristics of urban
Land use is a significant influence on nutrient export as controlled by impervious area, sanitary infrastructure, fertilizer application, and other determinants of input, retention, and stormwater transport. Tables 3-2A and 3-2B compare dissolved nitrate, total nitrogen, phosphate, and total phosphorus loads exported from forest catchments with catchments in different developed land uses studied by the Baltimore Ecosystem Study (Groffman et al., 2004). Loads were computed with the Fluxmaster system (Schwarz et al., 2006) from weekly samples taken at outlet gauges. In these sites in Baltimore County, the forested catchment, Pond Branch, has nitrogen loads one to two orders of magnitude lower than the developed catchments. Baisman Run, with one-third of the catchment in low-density, septic-served suburban land use, has nitrogen export exceeding Dead Run, an older, dense urban catchment. In this case, nutrient load does not follow the direct variation of impervious area because of the switch to septic systems and greater fertilizer use in lower density areas. However, Figure 3-26 shows that as impervious area increases, a much greater proportion of the total nitrogen load is discharged in less frequent, higher runoff events (Shields et al., 2008), reducing the potential to decrease loads by on-site SCMs. Total phosphorus loads were similarly as low (0.05–0.6 kg P/ha/yr) as nitrogen in the Pond Branch catchment (forest) over the 2000–2004 time period, and one to two orders of magnitude lower compared to agricultural and residential catchments.

It should be noted that specific areal loading rates, even in undeveloped catchments, can vary significantly depending on rates of atmospheric deposition, disturbance, and climate conditions. The hydrologic connectivity of nonpoint pollutant source areas to receiving waterbodies is also a critical control on loading in developed catchments (Nadeau and Rains, 2007) and is dependent on both properties of the pollutant as well as the catchment hydrology. For example, total nitrogen was high in both the agricultural and low-density suburban sites. Total phosphorus, on the other hand, was high in the Baltimore Ecosystem Study agricultural catchment, but close to the concentration of the forest site in the low-density suburban site serviced by septic systems. This is because septic systems tend to retain phosphorus, while septic wastewater nitrogen is typically nitrified in the unsaturated zone below a spreading field and efficiently transported in the groundwater to nearby streams.

### TABLE 3-2A Dissolved Nitrate and Total Nitrogen Export Rates from Forest and Developed Land-Use Catchments in the Baltimore Ecosystem Study

<table>
<thead>
<tr>
<th>Catchment</th>
<th>Land Use</th>
<th>Nitrate (kg N/ha/yr)</th>
<th>Total N (kg N/ha/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pond Branch</td>
<td>Forest</td>
<td>0.11</td>
<td>0.08</td>
</tr>
<tr>
<td>McDonogh</td>
<td>Agriculture</td>
<td>17.6</td>
<td>12.9</td>
</tr>
<tr>
<td>Baisman Run</td>
<td>Mixed Forest and Suburban</td>
<td>7.2</td>
<td>3.8</td>
</tr>
<tr>
<td>Dead Run</td>
<td>Urban</td>
<td>3.0</td>
<td>2.9</td>
</tr>
</tbody>
</table>
TABLE 3-2B  Dissolved Phosphate and Total Phosphorus Export Rates from Forest and Developed Land-Use Catchments in the Baltimore Ecosystem Study

<table>
<thead>
<tr>
<th>Catchment</th>
<th>Land Use</th>
<th>Phosphate (kg P/ha/yr)</th>
<th>Total P (kg P/ha/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pond Branch</td>
<td>Forest</td>
<td>0.009</td>
<td>0.007</td>
</tr>
<tr>
<td>McDonogh</td>
<td>Agriculture</td>
<td>0.12</td>
<td>0.080</td>
</tr>
<tr>
<td>Baisman Run</td>
<td>Mixed Forest and Suburban</td>
<td>0.009</td>
<td>0.005</td>
</tr>
<tr>
<td>Dead Run</td>
<td>Urban</td>
<td>0.039</td>
<td>0.037</td>
</tr>
</tbody>
</table>

FIGURE 3-26  Cumulative transport of total nitrogen at increasing flow levels from catchments in Baltimore City and County including dominantly forest (Pond Branch), low-density development on septic systems and forest (Baisman Run), agricultural (McDonogh), medium-density suburban development on separate sewers (Glyndon), and higher-density residential, commercial, and highway land cover (Dead Run). SOURCE: Reprinted, with permission, from Shields et al. (2008). Copyright 2008 by the American Geophysical Union.
TABLE 3-3  Relative Sources of Parameters of Concern for Different Land Uses in Urban Areas

<table>
<thead>
<tr>
<th>Problem Parameter</th>
<th>Residential</th>
<th>Commercial</th>
<th>Industrial</th>
<th>Freeway</th>
<th>Construction</th>
</tr>
</thead>
<tbody>
<tr>
<td>High flow rates (energy)</td>
<td>Low</td>
<td>High</td>
<td>Moderate</td>
<td>High</td>
<td>Moderate</td>
</tr>
<tr>
<td>Large runoff volumes</td>
<td>Low</td>
<td>High</td>
<td>Moderate</td>
<td>High</td>
<td>Moderate</td>
</tr>
<tr>
<td>Debris (floatables and gross solids)</td>
<td>High</td>
<td>High</td>
<td>Low</td>
<td>Moderate</td>
<td>High</td>
</tr>
<tr>
<td>Sediment</td>
<td>Low</td>
<td>Moderate</td>
<td>Low</td>
<td>Low</td>
<td>Very high</td>
</tr>
<tr>
<td>Inappropriate discharges (mostly sewage and cleaning wastes)</td>
<td>Moderate</td>
<td>High</td>
<td>Moderate</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td>Microorganisms</td>
<td>High</td>
<td>Moderate</td>
<td>Moderate</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td>Toxicants (heavy metals/organics)</td>
<td>Low</td>
<td>Moderate</td>
<td>High</td>
<td>High</td>
<td>Moderate</td>
</tr>
<tr>
<td>Nutrients (eutrophication)</td>
<td>Moderate</td>
<td>Moderate</td>
<td>Low</td>
<td>Low</td>
<td>Moderate</td>
</tr>
<tr>
<td>Organic debris (SOD and DO)</td>
<td>High</td>
<td>Low</td>
<td>Low</td>
<td>Low</td>
<td>Moderate</td>
</tr>
<tr>
<td>Heat (elevated water temperature)</td>
<td>Moderate</td>
<td>High</td>
<td>Moderate</td>
<td>High</td>
<td>Low</td>
</tr>
</tbody>
</table>

NOTE: SOD, sediment oxygen demand; DO, dissolved oxygen.

runoff are controlled largely by the increase in volume and the washoff of pollutants from impervious surfaces. Stormwater in this phase is associated with increases in discharges of most pollutants, but with less sediment washoff than from construction and likely less sediment and nutrient discharges compared to any pre-urbanization agricultural operations (although increased channel erosion may increase the mass of sediment delivered in this phase; Pitt et al., 2007). A third significant urban land use is industrial activity. As described later, industrial site stormwater discharges are highly variable, but often greater than other land uses.

Construction Site Erosion Characteristics

Problems associated with construction site runoff have been known for many years. More than 25 years ago, Willett (1980) estimated that approximately 5 billion tons of sediment reached U.S. surface waters annually, of which 30 percent was generated by natural processes and 70 percent by human activities. Half of this 70 percent was attributed to eroding croplands. Although construction occurred on only about 0.007 percent of U.S. land in the 1970s, it ac-
counted for approximately 10 percent of the sediment load to all U.S. surface waters and equaled the combined sediment contributions of forestry, mining, industrial, and commercial land uses (Willett, 1980).

Construction accounts for a much greater proportion of the sediment load in urban areas than it does in the nation as a whole. This is because construction sites have extremely high erosion rates and because urban construction sites are efficiently drained by stormwater drainage systems installed early during the construction activities. Construction site erosion losses vary greatly throughout the nation, depending on local rain, soil, topographic, and management conditions. As an example, the Birmingham, Alabama, area may have some of the highest erosion rates in the United States because of its combination of very high-energy rains, moderately to severely erosive soils, and steep slopes (Pitt et al., 2007). The typically high erosion rates mean that even a small construction project may have a significant detrimental effect on local waterbodies.

Extensive evaluations of urban construction site runoff problems have been conducted in Wisconsin for many years. Data from the highly urbanized Menomonee River watershed in southeastern Wisconsin indicate that construction sites have much greater potentials for generating sediment and phosphorus than do other land uses (Chesters et al., 1979). For example, construction sites can generate approximately 8 times more sediment and 18 times more phosphorus than industrial sites (the land use that contributes the second highest amount of these pollutants) and 25 times more sediment and phosphorus than row crops. In fact, construction sites contributed more sediment and phosphorus to the Menomonee River than any other land use, although in 1979, construction comprised only 3.3 percent of the watershed’s total land area. During this early study, construction sites were found to contribute about 50 percent of the suspended sediment and total phosphorus loading at the river mouth (Novotny and Chesters, 1981).

Similar conclusions were reported by the Southeastern Wisconsin Regional Planning Commission (SEWRPC) in a 1978 modeling study of the relative pollutant contributions of 17 categories of point and nonpoint pollution sources to 14 watersheds in the southeast Wisconsin regional planning area (SEWRPC, 1978). This study revealed construction as the first or second largest contributor of sediment and phosphorus in 12 of the 14 watersheds. Although construction occupied only 2 percent of the region’s total land area in 1978, it contributed approximately 36 percent of the sediment and 28 percent of the total phosphorus load to inland waters, making construction the region’s second largest source of these two pollutants. The largest source of sediment was estimated to be cropland; livestock operations were estimated to be the largest source of phosphorus. By comparison, cropland comprised 72 percent of the region’s land area and contributed about 45 percent of the sediment and only 11 percent of the phosphorus to regional watersheds. When looking at the Milwaukee River watershed as a whole, construction is a major sediment contributor, even though the amount of land under active construction is very low. Construction areas were estimated to contribute about 53 percent of the total sediment discharged by the
Milwaukee River in 1985 (total sediment load of 12,500 lb/yr), while croplands contributed 25 percent, streambank erosion contributed 13 percent, and urban runoff contributed 8 percent.

Line and White (2007) recently investigated runoff characteristics from two similar drainage areas in the Piedmont region of North Carolina. One of the drainage areas was being developed as part of a large residential subdivision during the course of the study, while the other remained forested or in agricultural fields. Runoff volume was 68 percent greater for the developing compared with the undeveloped area, and baseflow as a percentage of overall discharge was approximately zero compared with 25 percent for the undeveloped area. Overall annual export of sediment was 95 percent greater for the developing area, while export of nitrogen and phosphorus forms was 66 to 88 percent greater for the developing area.

The biological stream impact of construction site runoff can be severe. For example, Hunt and Grow (2001) describe a field study conducted to determine the impact to a stream from a poorly controlled construction site, with impact being measured via fish electroshocking and using the Qualitative Habitat Evaluation Index. The 33-acre construction site consisted of severely eroded silt and clay loam subsoil and was located within the Turkey Creek drainage, Scioto County, Ohio. The number of fish species declined (from 26 to 19) and the number of fish found decreased (from 525 to 230) when comparing upstream unimpacted reaches to areas below the heavily eroding site. The Index of Biotic Integrity and the Modified Index of Well-Being, common fisheries indexes for stream quality, were reduced from 46 to 32 and 8.3 to 6.3, respectively. Upstream of the area of impact, Turkey Creek had the highest water quality designation available, but fell to the lowest water quality designation in the area of the construction activity. Water quality sampling conducted at upstream and downstream sites verified that the decline in fish diversity was not due to chemical affects alone.

Municipal Stormwater Characteristics

The suite of stormwater pollutants generated by municipal areas is expected to be much more diverse than construction sites because of the greater variety of land uses and pollutant source areas found within a typical city. Many studies have investigated stormwater quality, with the U.S. Environmental Protection Agency’s (EPA’s) NURP (EPA, 1983) being the best known and earliest effort to collect and summarize these data. Unfortunately, NURP was limited in that it did not represent all areas of the United States or all important land uses. More recently, the National Stormwater Quality Database (NSQD) (CWP and Pitt, 2008; Pitt et al., 2008 for version 3) has been compiling data from the EPA’s NPDES stormwater permit program for larger Phase I municipal separate storm sewer system (MS4) communities. As a condition of their Phase I permits, municipalities were required to establish a monitoring program to characterize their local stormwater quality for their most important land uses discharging to the
EFFECTS OF URBANIZATION ON WATERSHEDS

MS4. Although only a few samples from a few locations were required to be monitored each year in each community, the many years of sampling and large number of communities has produced a database containing runoff quality information for nearly 8,000 individual storm events over a wide range of urban land uses. The NSQD makes it possible to statistically compare runoff from different land uses for different areas of the country.

A number of land uses are represented in MS4 permits and also the database, including industrial stormwater discharges to an MS4. However, there is no separate compilation of quantitative mass emissions from specific industrial stormwater sources that may have been collected under industrial permit monitoring efforts. The observations in the NSQD were all obtained at outfall locations and do not include snowmelt or construction erosion sources. The most recent version of the NSQD contains stormwater data from about one-fourth of the total number of communities that participated in the Phase I NPDES stormwater permit monitoring activities. The database is located at http://unix.eng.ua.edu/~rpitt/Research/ms4/mainms4.shtml.

Table 3-4 is a summary of some of the stormwater data included in NSQD version 3, while Figure 3-27 shows selected plots of these data. The table describes the total number of observations, the percentage of observations above the detection limits, the median, and coefficients of variation for a few of the major constituents for residential, commercial, industrial, institutional, freeway, and open-space land-use categories, although relatively few data are available for institutional and open-space areas. It should be noted that even if there are significant differences in the median concentrations by the land uses, the range of the concentrations within single land uses can still be quite large. Furthermore, plots like Figure 3-27 do not capture the large variability in data points observed at an individual site.

There are many factors that can be considered when examining the quality of stormwater, including land use, geographical region, and season. The following is a narrative summary of the entire database and may not reflect information in Table 3-4 and Figure 3-29, which show only subsets of the data. First, statistical analyses of variance on the NSQD found significant differences among land-use categories for all of the conventional constituents, except for dissolved oxygen. (Turbidity, total solids, total coliforms, and total E. coli did not have enough samples in each group to evaluate land-use differences.) Freeway sites were found to be significant sources of several pollutants. For example, the highest TSS, COD, and oil and grease concentrations (but not necessarily the highest median concentrations) were reported for freeways. The median ammonia concentration in freeway stormwater is almost three times the median concentration observed in residential and open-space land uses, while freeways have the lowest orthophosphate and nitrite–nitrate concentrations—half of the concentration levels that were observed in industrial land uses.

In almost all cases the median metal concentrations at the industrial areas were about three times the median concentrations observed in open-space and residential areas. The highest lead and zinc concentrations (but not necessarily
### TABLE 3-4  Summary of Selected Stormwater Quality Data Included in NSQD, Version 3.0

<table>
<thead>
<tr>
<th></th>
<th>TSS (mg/L)</th>
<th>COD (mg/L)</th>
<th>Fecal Coli. (mpn/100 mL)</th>
<th>Nitrogen. Total Kjeldahl (mg/L)</th>
<th>Phosphorus. Total (mg/L)</th>
<th>Ca. Total (µg/L)</th>
<th>Pb. Total (µg/L)</th>
<th>Zn. Total (µg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>All Areas Combined (8,139)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Coefficient of variation (COV)</td>
<td>2.2</td>
<td>1.1</td>
<td>5.0</td>
<td>1.2</td>
<td>2.0</td>
<td>1.1</td>
<td>2.0</td>
<td>3.3</td>
</tr>
<tr>
<td>Median</td>
<td>82.0</td>
<td>53.0</td>
<td>4300</td>
<td>1.3</td>
<td>0.2</td>
<td>1.0</td>
<td>1.0</td>
<td>1.0</td>
</tr>
<tr>
<td>Number of samples</td>
<td>6786</td>
<td>5870</td>
<td>2154</td>
<td>6156</td>
<td>7425</td>
<td>5165</td>
<td>4884</td>
<td>6164</td>
</tr>
<tr>
<td>% samples above detection</td>
<td>59</td>
<td>99</td>
<td>91</td>
<td>97</td>
<td>97</td>
<td>89</td>
<td>78</td>
<td>98</td>
</tr>
<tr>
<td><strong>All Residential Areas Combined (2,696)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>COV</td>
<td>2.0</td>
<td>1.0</td>
<td>5.7</td>
<td>1.2</td>
<td>1.6</td>
<td>1.9</td>
<td>2.1</td>
<td>3.3</td>
</tr>
<tr>
<td>Median</td>
<td>82.0</td>
<td>53.0</td>
<td>4300</td>
<td>1.3</td>
<td>0.3</td>
<td>11.0</td>
<td>8.0</td>
<td>7.0</td>
</tr>
<tr>
<td>Number of samples</td>
<td>2167</td>
<td>1473</td>
<td>636</td>
<td>203</td>
<td>2268</td>
<td>1643</td>
<td>1279</td>
<td>1912</td>
</tr>
<tr>
<td>% samples above detection</td>
<td>59</td>
<td>99</td>
<td>86</td>
<td>98</td>
<td>98</td>
<td>88</td>
<td>77</td>
<td>97</td>
</tr>
<tr>
<td><strong>All Commercial Areas Combined (915)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>COV</td>
<td>1.7</td>
<td>1.0</td>
<td>3.0</td>
<td>0.9</td>
<td>1.2</td>
<td>1.4</td>
<td>1.7</td>
<td>1.4</td>
</tr>
<tr>
<td>Median</td>
<td>55.0</td>
<td>63.0</td>
<td>3000</td>
<td>1.3</td>
<td>0.2</td>
<td>15.0</td>
<td>15.0</td>
<td>110.0</td>
</tr>
<tr>
<td>Number of samples</td>
<td>643</td>
<td>640</td>
<td>276</td>
<td>726</td>
<td>500</td>
<td>795</td>
<td>865</td>
<td>839</td>
</tr>
<tr>
<td>% samples above detection</td>
<td>97</td>
<td>98</td>
<td>86</td>
<td>98</td>
<td>95</td>
<td>85</td>
<td>79</td>
<td>90</td>
</tr>
<tr>
<td><strong>All Industrial Areas Combined (719)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>COV</td>
<td>1.7</td>
<td>1.0</td>
<td>6.1</td>
<td>1.1</td>
<td>1.4</td>
<td>1.1</td>
<td>2.0</td>
<td>1.7</td>
</tr>
<tr>
<td>Median</td>
<td>73.0</td>
<td>59.0</td>
<td>2850</td>
<td>1.4</td>
<td>0.2</td>
<td>11.0</td>
<td>10.0</td>
<td>15.0</td>
</tr>
<tr>
<td>Number of samples</td>
<td>594</td>
<td>474</td>
<td>217</td>
<td>569</td>
<td>606</td>
<td>530</td>
<td>960</td>
<td>596</td>
</tr>
<tr>
<td>% samples above detection</td>
<td>98</td>
<td>98</td>
<td>94</td>
<td>97</td>
<td>95</td>
<td>80</td>
<td>76</td>
<td>99</td>
</tr>
</tbody>
</table>
### EFFECTS OF URBANIZATION ON WATERSHEDS


<table>
<thead>
<tr>
<th>Dataset Description</th>
<th>COV</th>
<th>Median</th>
<th>Number of samples</th>
<th>% samples above detection</th>
</tr>
</thead>
<tbody>
<tr>
<td>All Freeway Areas Combined (680)</td>
<td>2.6</td>
<td>64.0</td>
<td>2600</td>
<td>1.7</td>
</tr>
<tr>
<td>All Institutional Areas Combined (24)</td>
<td>1.1</td>
<td>1.0</td>
<td>94</td>
<td>0.6</td>
</tr>
<tr>
<td>All Open-Space Areas Combined (79)</td>
<td>1.8</td>
<td>12.3</td>
<td>2300</td>
<td>0.4</td>
</tr>
</tbody>
</table>

NOTE: COV and Median values are provided as an example. The actual values may vary.

- COV: Coefficient of Variation
- Median: This represents the middle value in a set of data
- Number of samples: The total number of samples in each dataset
- % samples above detection: This indicates the percentage of samples that exceed a certain threshold.
FIGURE 3-27  Grouped box and whisker plots of data from the NSQD. The median values are indicated with the horizontal line in the center of the box, while the ends of the box represent the 25th and 75th percentile values. The whiskers extend to the 5th and 95th percentile values, and values outside of these extremes are indicated with separate dots. These groups were statistically analyzed and were found to have at least one group that is significantly different from the other groups. The ranges of the values in each group are large, but a very large number of data points is available for each group. The grouping of the data into these categories helps explain much of the total variability observed, and the large number of samples in each category allows suitable statistical tests to be made. Many detailed analyses are presented at the NSQD website (Maestre and Pitt, 2005).
EFFECTS OF URBANIZATION ON WATERSHEDS  187

the highest median concentrations) were found in industrial land uses. Lower concentrations of TDS, five-day biological oxygen demand (BOD₅), and fecal coliforms were observed in industrial land-use areas. By contrast, the highest concentrations of dissolved and total phosphorus were associated with residential land uses. Fecal coliform concentrations are also relatively high for residential and mixed residential land uses. Open-space land-use areas show consistently low concentrations for the constituents examined. There was no significant difference noted for total nitrogen among any of the land uses monitored.

In terms of regional differences, significantly higher concentrations of TSS, BOD₅, COD, total phosphorus, total copper, and total zinc were observed in arid and semi-arid regions compared to more humid regions. In contrast, fecal coliforms and total dissolved solids were found to be higher in the upper Midwest. More detailed discussions of land use and regional differences in stormwater quality can be found in Maestre et al. (2004) and Maestre and Pitt (2005, 2006). In addition to the information presented above, numerous researchers have conducted source area monitoring to characterize sheet flows originating from urban surfaces (such as roofs, parking lots, streets, landscaped areas, storage areas, and loading docks). The reader is referred to Pitt et al. (2005a,b,c) for much of this information.

Industrial Stormwater Characteristics

The NSQD, described earlier, has shown that industrial-area stormwater has higher concentrations of most pollutants compared to other land uses, although the variability is high. MS4 monitoring activities are usually conducted at outfalls of drainage systems containing many individual industrial activities, so discharge characteristics for specific industrial types are rarely available. This discussion provides some additional information concerning industrial stormwater beyond that included in the previous discussion of municipal stormwater. In general, there is a profound lack of data on industrial stormwater compared to municipal stormwater, and a correspondingly greater uncertainty about industrial stormwater characteristics.

The first comprehensive monitoring of an industrial area that included stormwater, dry weather base flows, and snowmelt runoff was conducted in selected Humber River catchments in Ontario (Pitt and McLean, 1986). Table 3-5 shows the annual mass discharges from the monitored industrial area in North York, along with ratios of these annual discharges compared to discharges from a mixed commercial and residential area in Etobicoke. The mass discharges of heavy metals, total phosphorus, and COD from industrial stormwater are three to six times that of the mixed residential and commercial areas.

Hotspots of contamination on industrial sites are a specific concern. Stormwater runoff from “hotspots” may contain loadings of hydrocarbons, trace metals, nutrients, pathogens and/or other toxicants that are greater than the loadings of “normal” runoff. Examples of these hotspots include airport de-icing
facilities, auto recyclers/junkyards, commercial garden nurseries, parking lots, vehicle fueling and maintenance stations, bus or truck (fleet) storage areas, industrial rooftops, marinas, outdoor transfer facilities, public works storage areas, and vehicle and equipment washing/steam cleaning facilities (Bannerman et al., 1993; Pitt et al., 1995; Claytor and Schueler, 1996).

The elevated concentrations and mass discharges found in stormwater at industrial sites are associated with both the activities that occur and the materials used in industrial areas, as discussed in the sections that follow.

**Effects of Roofing Materials on Stormwater Quality**

The extensive rooftops of industrial areas can be a significant pollutant source area. A summary of the literature on roof-top runoff quality, including both roof surfaces and underlying materials used as subbases (such as treated wood), is presented in Table 3-6. Good (1993) found that dissolved metals' concentrations and toxicity remained high in roof runoff samples, especially from rusty galvanized metal roofs during both first flush and several hours after a rain has started, indicating that metal leaching continued throughout the events and for many years. During pilot-scale tests of roof panels exposed to rains over a two-year period, Clark et al. (2008) found that copper roof runoff concentrations for newly treated wood panels exceeded 5 mg/L (a very high value compared to median NSQD stormwater concentrations of about 10 to 40 µg/L for different land uses) for the first nine months of exposure. These results indicated that copper continued to be released from these wood products at levels high enough to exceed aquatic life criteria for long periods after installation, and were not simply due to excess surface coating washing off in the first few storms after installation.
<table>
<thead>
<tr>
<th>Roof Type</th>
<th>Location</th>
<th>Water Quality Parameter</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Polyester Tiled Flat gravel</td>
<td>Duebendorf, Switzerland</td>
<td>Cu (µg/L): 6817/1905</td>
<td>Boiler</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Zn (µg/L): 140/36</td>
<td>(1997)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pb (µg/L): 2076/36</td>
<td></td>
</tr>
<tr>
<td>Plywood w/ roof paper/ter</td>
<td>Washington</td>
<td>Cu (µg/L): 166/128</td>
<td>Good</td>
</tr>
<tr>
<td>Rusty galvanized metal</td>
<td></td>
<td>Zn (µg/L): 905/200</td>
<td>(1993)</td>
</tr>
<tr>
<td>Old metal w/Al paint</td>
<td></td>
<td>Pb (µg/L): 12200/1190</td>
<td></td>
</tr>
<tr>
<td>Flat tar surface w/lithium</td>
<td></td>
<td>Cd (µg/L): 11/30</td>
<td></td>
</tr>
<tr>
<td>reflective Al paint</td>
<td></td>
<td>As (µg/L): 1980/1610</td>
<td></td>
</tr>
<tr>
<td>New anodized Al</td>
<td></td>
<td>pH: 297/257</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Zn (µg/L): 5901/µg/g</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pb (µg/L): 670 µg/g</td>
<td></td>
</tr>
<tr>
<td>Concrete slate tiles</td>
<td></td>
<td>Zn (µg/L): 7.75</td>
<td></td>
</tr>
<tr>
<td>Asbestos cement sheets</td>
<td></td>
<td>Pb (µg/L): 7.09</td>
<td></td>
</tr>
<tr>
<td>Aluminum sheets</td>
<td></td>
<td>Cd (µg/L): 6.68</td>
<td></td>
</tr>
<tr>
<td>Cu panels</td>
<td>Munich, Germany</td>
<td>As (µg/L): 6.7–7.0</td>
<td>Athanasiadi s. et al. (2006)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>ND</td>
<td></td>
</tr>
<tr>
<td>CCA wood</td>
<td>Florida</td>
<td>Mg (mg/L): 1200–1800</td>
<td>Khan et al. (2006)</td>
</tr>
<tr>
<td>Untreated wood</td>
<td></td>
<td>Ca (mg/L): 2–3</td>
<td></td>
</tr>
</tbody>
</table>

Note: D, dissolved; T, total; ND, not detected.

SOURCE: Reprinted, with permission, from Clark et al. (2008). Copyright 2003 by American Society of Civil Engineers.
Traditional unpainted or uncoated hot-dip galvanized steel roof surfaces can also produce very high zinc concentrations. For example, pilot-scale tests by Clark et al. (2008) indicated that zinc roof runoff concentrations were 5 to 30 mg/L throughout the first two years of monitoring of a traditional galvanized metal panel. These are very high values compared to median stormwater values reported in the NSQD of 60 to 300 µg/L for different land uses. Factory-painted aluminum–zinc alloy panels had runoff zinc levels less than 250 µg/L, which were closer to the reported NSQD median values. The authors concluded that traditional galvanized metal roofing contributed the greatest concentrations of many metals and nutrients. In addition, they found that pressure-treated and waterproofed wood contributed substantial copper loads. The potential for nutrient release exists in many of the materials tested (possibly as a result of phosphate washes and binders used in the material’s preparation or due to natural degradation).

Other researchers have investigated the effects of industrial rooftop runoff on receiving waters and biota. Bailey et al. (1999) investigated the toxicity to juvenile rainbow trout of runoff from British Columbia sawmills and found that much of the toxicity may have been a result of divalent cations on the industrial site, especially zinc from galvanized roofs.

Effects of Pavement and Pavement Maintenance on Stormwater Quality

Pavement surfaces can also have a strong influence on stormwater runoff quality. For example, concrete is often mixed with industrial waste sludges as a way of disposing of the wastes. However, this can lead to stormwater discharges high in toxic compounds, either due to the additives themselves or due to the mobilization of compounds via the additives. Salaita and Tate (1998) showed that high levels of aluminum, iron, calcium, magnesium, silicon, and sodium were seen in the cement-waste samples. A variety of sands, including waste sands, have been suggested as potential additives to cement and for use as fill in roadway construction. Wiebusch et al. (1998) tested brick sands and found that the higher the concentration of alkaline and alkaline earth metals in the samples, the more easily the heavy metals were released. Pitt et al. (1995) also found that concrete yard runoff had the highest toxicity (using Microtox screening methods) observed from many source areas, likely due to the elevated pH (about 11) from the lime dust washing off from the site.

The components of asphalt have been investigated by Rogge et al. (1997), who found that the majority of the elutable organic mass that could be identified consisted of \( n \)-alkanes (73 percent), carboxylic acids such as \( n \)-alkanoic acids (17 percent), and benzoic acids. PAHs and thiaarenes were 7.9 percent of the identifiable mass. In addition, heterocyclic aromatic hydrocarbons containing sulfur (S-PAH), such as dibenzothiophene, were identified at concentration lev-
els similar to that of phenanthrene. S-PAHs are potentially mutagenic (similar to other PAHs), but due to their slightly increased polarity, they are more soluble in water and more prone to aquatic bioaccumulation.

In addition to the bitumens and asphalts, other compounds are added to paving (and asphaltic roofing) materials. Chemical modifiers are used both to increase the temperature range at which asphalts can be used and to prevent stripping of the asphalt from the binder. A variety of fillers may also be used in asphalt pavement mixtures. The long-term environmental effects of these chemicals in asphalts are unknown. Reclaimed asphalt pavements have also been proposed for use as fill materials for roadways. Brantley and Townsend (1999) performed a series of leaching tests and analyzed the leachate for a variety of organics and heavy metals. Only lead from asphalt pavements reclaimed from older roadways was found to be elevated in the leachate.

Stormwater quality from asphalt-paved surfaces seems to vary with time. Fish kills have been reported when rains occur shortly after asphalt has been installed in parking areas near ponds or streams (Anonymous, 2000; Perez-Rivas, 2000; Kline, 2002). It is expected that these effects are associated with losses of the more volatile and toxic hydrocarbons that are present on new surfaces. It is likely that the concentrations of these materials in runoff decrease as the pavement ages. Toxicity tests conducted on pavements several years old have not indicated any significant detrimental effects, except for those associated with activities conducted on the surface (such as maintenance and storage of heavy equipment; Pitt et al., 1995, 1999). However, pavement maintenance used to "renew" the asphalt surfaces has been shown to cause significant problems, which are summarized below.

A significant source of PAHs in the Austin, Texas, area (and likely elsewhere) has been identified as coal-tar sealants commonly used to "restore" asphalt parking lots and storage areas. Mahler et al. (2005) found that small particles of sealcoat that flake off due to abrasion by vehicle tires have PAH concentrations about 65 times higher than for particles washed off parking lots that are not seal coated. Unsealed parking lots receive PAHs from the same urban sources as do sealed parking lots (e.g., tire particles, leaking motor oil, vehicle exhaust, and atmospheric fallout), and yet the average yield of PAHs from the sealed parking lots was found to be 50 times greater than that from the control lots. The authors concluded that sealed parking lots could be the dominant source of PAHs in watersheds that have seal-coated surfaces, such as many industrial, commercial, and residential areas. Consequently, the City of Austin has restricted the use of parking lot coal-tar sealants, as have several Wisconsin communities.

Stored Materials Exposed to Rain

Although roofing and pavement materials make up a large fraction of the total surface covers and can have significant effects on stormwater quality, leaching of rain through stored materials may also be a significant pollutant
source at industrial sites. Exposed metals in scrap yards can result in very high concentrations of heavy metals. For example, Table 3-7 summarizes data from three metals recycling facilities/scrap yards in Wisconsin and shows the large fraction of metals that are either dissolved in the runoff or associated with very fine particulate matter. For most of these metals, their greatest abundance is associated with the small particles (<20 µm in diameter), and relatively little is associated with the filterable fraction. These metals concentrations (especially zinc, copper, and lead) are also very high compared to that of most outfall industrial stormwater.

**OTHER SOURCES OF URBAN RUNOFF DISCHARGES**

Wet weather stormwater discharges from separate storm sewer outfalls are not the only discharges entering receiving waters from these systems. Dry weather flows, snowmelt, and atmospheric deposition all contribute to the pollutant loading of urban areas to receiving waters, and for some compounds may be the largest contributor. Many structural SCMs, especially those that rely on sedimentation or filtration, have been designed to function primarily with stormwater and are not nearly as effective for dry weather discharges, snowmelt, or atmospheric deposition because these nontraditional sources vary considerably in key characteristics, such as the flow rate and volume to be treated, sediment concentrations and particle size distribution, major competing ions, association of pollutants with particulates of different sizes, and temperature. Information on the treatability of stormwater vs. snowmelt and other nontraditional sources of urban runoff can be found in Pitt and McLean (1986), Pitt et al. (1995), Johnson et al. (2003), and Morquecho (2005).

**TABLE 3-7  Metal Concentration Ranges Observed in Scrapyard Runoff**

<table>
<thead>
<tr>
<th>Particle Size</th>
<th>Iron (mg/L)</th>
<th>Aluminum (mg/L)</th>
<th>Zinc (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total</td>
<td>20 – 810</td>
<td>15 – 70</td>
<td>1.6 – 8</td>
</tr>
<tr>
<td>&lt; 63 µm diameter</td>
<td>22 – 767</td>
<td>15 – 58</td>
<td>1.5 – 7.6</td>
</tr>
<tr>
<td>&lt; 38 µm diameter</td>
<td>21 – 705</td>
<td>15 – 58</td>
<td>1.4 – 7.4</td>
</tr>
<tr>
<td>&lt; 20 µm diameter</td>
<td>15 – 534</td>
<td>12 – 50</td>
<td>1.1 – 7.2</td>
</tr>
<tr>
<td>&lt; 0.45 µm diameter</td>
<td>0.1 – 38</td>
<td>0.1 – 5</td>
<td>0.1 – 6.7</td>
</tr>
<tr>
<td>(filterable fraction)</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Copper (mg/L)</th>
<th>Lead (mg/L)</th>
<th>Chromium (mg/L)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total</td>
<td>1.1 – 3.8</td>
<td>0.6 – 1.7</td>
</tr>
<tr>
<td>&lt; 63 µm diameter</td>
<td>1.1 – 3.6</td>
<td>0.1 – 1.6</td>
</tr>
<tr>
<td>&lt; 38 µm diameter</td>
<td>1.1 – 3.3</td>
<td>0.1 – 1.6</td>
</tr>
<tr>
<td>&lt; 20 µm diameter</td>
<td>1.0 – 2.8</td>
<td>0.1 – 1.6</td>
</tr>
<tr>
<td>&lt; 0.45 µm diameter</td>
<td>0.1 – 0.3</td>
<td>0.1 – 0.3</td>
</tr>
<tr>
<td>(filterable fraction)</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*SOURCE: Reprinted, with permission, from Clark et al. (2000). Copyright 2000 by Shirley Clark.*
Dry Weather Flows

At many stormwater outfalls, discharges occur during dry weather. These may be associated with discharges from leaking sanitary sewer and drinking water distribution systems, industrial wastewaters, irrigation return flows, or natural spring water entering the system (Figures 3-28 to 3-33). Possibly 25 percent of all separate stormwater outfalls have water flowing in them during dry weather, and as much as 10 percent are grossly contaminated with raw sewage, industrial wastewaters, and so forth (Pitt et al., 1993). These flow contributions can be significant on an annual mass basis, even though the flow rates are relatively small, because they have long duration. This is particularly true in arid areas, where dry weather discharges can occur daily. For example, despite the fact that rain is scarce from May to September in Southern California, an estimated 40 to 90 million liters of discharge flow per day into Santa Monica Bay through approximately 70 stormwater outlets that empty onto or across beaches (LAC DPW, 1985; SMBRP, 1994), such that the contribution of dry weather flow to the total volume of runoff into the bay is about 30 percent (NRC, 1984). Furthermore, in the nearby Ballona Creek watershed, dry weather discharges of trace metals were found to comprise from 8 to 42 percent of the total annual loading (McPherson et al., 2002). Stein and Tiefenthaler (2003) further found that the highest loadings of metals and bacteria in this watershed discharging during dry weather can be attributed to a few specific stormwater drains.

In many cases, stormwater managers tend to overlook the contribution of dry weather discharges, although the EPA’s NPDES Stormwater Permit program requires municipalities to conduct stormwater outfall surveys to identify, and then correct, inappropriate discharges into separate storm sewer systems. The role of inappropriate discharges in the NPDES Stormwater Permit program, the developed and tested program to identify and quantify their discharges, and an extensive review of these programs throughout the United States can be found in the recently updated report prepared for the EPA (CWP and Pitt, 2004).
FIGURE 3-29  Contamination of storm drainage with inappropriate disposal of oil.  
SOURCE: Courtesy of the Center for Watershed Protection.

FIGURE 3-30  Dry weather flows from Toronto industrial area outfall.  SOURCE: Pitt and McLean (1986).
FIGURE 3-31 Sewage from clogged system overflowing into storm drainage system. SOURCE: Robert Pitt, University of Alabama.

FIGURE 3-32 Failing sanitary sewer, causing upwelling of sewage through soil, and draining to gutter and then to storm drainage system. SOURCE: Robert Pitt, University of Alabama.
In northern areas, snowmelt runoff can be a significant contributor to the annual discharges from urban areas through the storm drainage system (see Figure 3-34). In locations having long and harsh winters, with little snowmelt until the spring, pollutants can accumulate and be trapped in the snowpack all winter until the major thaw when the contaminants are transported in short-duration events to the outfalls (Jokela, 1990). The sources of the contaminants accumulating in snowpack depend on the location, but they usually include emissions from nearby motor vehicles and heating equipment and industrial activity in the neighborhood. Dry deposition of sulfur dioxide from industrial and power plant smokestacks affects snow packs over a wider area and has frequently been studied because of its role in the acid deposition process (Cadle, 1991). Pollutants are also directly deposited on the snowpack. The sources of directly deposited pollutants include debris from deteriorated roadways, vehicles depositing petroleum products and metals, and roadway maintenance crews applying salt and anti-skid grit (Oberts, 1994). Urban snowmelt, like rain runoff, washes some material off streets, roofs, parking and industrial storage lots, and drainage gutters. However, snowmelt runoff usually has much less energy than striking rain and heavy flowing stormwater. Novotny et al. (1986) found that urban soil ero-
Erosion is reduced or eliminated during winter snow-cover conditions. However, erosion of bare ground at construction sites in the spring due to snowmelt can still be very high (Figure 3-35).

FIGURE 3-34 Snowmelt photos. SOURCE: Roger Bannerman, Wisconsin Department of Natural Resources.

FIGURE 3-35 Construction site in early spring after snowmelt showing extensive sediment transport. SOURCE: Roger Bannerman, Wisconsin Department of Natural Resources.
Sources of Contaminants in Snowmelt

Several mechanisms can bring about contamination of snow and snowmelt waters. Initially, air pollutants can be incorporated into snowflakes as they form and fall to the ground. After it falls to the ground and accumulates, the snow can become further contaminated by dry atmospheric deposition, deposition of nearby lost fugitive dust materials (usually blown onto snow packs near roads by passing vehicles), and wash off of particulates from the exposed ground surfaces as it melts and flows to the drainage system.

Snowflakes can remove particulates and gases from the air by in-cloud or below-cloud capture. In-cloud capture of pollutants can occur during snowflake formation as super-cooled cloud water condenses on particles and aerosols that act as cloud condensation nuclei. This is known as nucleation scavenging and is a major pathway for air pollution to be incorporated into snow. Particles and gases may also be scavenged as snowflakes fall to the ground. Gases can also be absorbed as snow falls. Snowflakes are more effective below-cloud scavengers than raindrops because they are bigger and fall slower. Barrie (1991) reports that large snowflakes capture particles in the 0.2- to 0.4-μm-diameter range, not by impaction but by filtering the air that moves through the snowflakes as they fall to the ground.

Most of the contamination of snow in urban areas likely occurs after it lands on the ground. Table 3-8 shows the flow-weighted mean concentrations of pollutants found in undisturbed falling snow compared to snow found in urban snow cover (Bennett et al., 1981). Pitt and McLean (1986) also measured snowpack contamination as a function of distance from a heavily traveled road passing through a park. The contaminants in the snow were at much greater concentrations near the road (the major source of blown contamination on the snow) than farther away. (The pollutant levels in the fresh fallen snow are generally a small fraction of the levels in the snow collected from urban study areas.) Pierstorff and Bishop (1980) also analyzed freshly fallen snow and compared the quality to snow stored at a snow dump site. They concluded that “pollutant levels at the dump site are the result of environmental input occurring after the snow falls.” Some pollutants in snowmelt have almost no atmospheric

<table>
<thead>
<tr>
<th></th>
<th>Fresh Fallen</th>
<th>High Density Land Use</th>
<th>Low Density Land Use</th>
</tr>
</thead>
<tbody>
<tr>
<td>COD</td>
<td>10</td>
<td>402</td>
<td>54</td>
</tr>
<tr>
<td>TS</td>
<td>86</td>
<td>2000</td>
<td>165</td>
</tr>
<tr>
<td>SS</td>
<td>16</td>
<td>545</td>
<td>4.5</td>
</tr>
<tr>
<td>TKN</td>
<td>0.19</td>
<td>2.69</td>
<td>2</td>
</tr>
<tr>
<td>NO₃</td>
<td>0.15</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>P</td>
<td>—</td>
<td>0.66</td>
<td>0.017</td>
</tr>
<tr>
<td>Pb</td>
<td>—</td>
<td>0.95</td>
<td>—</td>
</tr>
</tbody>
</table>

Note: The units are mg/L. SOURCE: Reprinted, with permission, from Bennett et al. (1981). Copyright 1981 by Water Pollution Control Federation.
sources. For example, Oliver et al. (1974) found negligible amounts of chlorides in samples of snow from rooftops, indicating that the high chloride level found in the snowmelt runoff water comes almost entirely from surface sources (i.e., road salting). Similar roadside snowpack observations along city park roads by Pitt and McLean (1986) also indicated the strong association of road salt with snowpack chloride levels.

Runoff and Pollutant Loading from Snowmelt

Snowmelt events can exhibit a first flush, in which there are higher concentrations of contaminants at the beginning compared to the total event averaged concentration. The enrichment of the first portion of a snowmelt event by soluble pollutants may be due to snowpack density changes, where water percolation and melt/freeze events that occur in the snowpack cause soluble pollutants to be flushed from throughout the snowpack to concentrate at the bottom of the pack (Colbeck, 1981). This concentrated layer leaves the snowpack as a highly concentrated pulse, as snow melts from the bottom due to warmth from the ground (Oberts, 1994).

When it rains on snow, heavy pollutant loads can be produced because both soluble and particulate pollutants are melted from the snowpack simultaneously. Also, the large volume of melt plus rain can wash off pollutants that have accumulated on various surfaces such as roads, parking lots, roofs, and saturated soil surfaces. The intensity of runoff from a rain-on-snow event can be greater than a summer thunderstorm because the ground is saturated or frozen and the rapidly melting snowpack provides added runoff volume (Oberts, 1994).

Figure 3-36 compares the runoff volumes associated with snowmelts alone to those associated with snowmelts mixed with rain from monitoring at an industrial area in Toronto (Pitt and McLean, 1986). Rain with snowmelt contributes over 80 percent of the total cold-weather event runoff volume.

![Figure 3-36](image_url)
Whether pollutant loadings are higher or lower for snowmelt than for rainfall depends on the particular pollutant and its seasonal prevalence in the environment. For example, the high concentrations of dissolved solids found in snowmelt are usually caused by high chloride concentrations that stem from the amount of de-icing salt used. Figure 3-37 is a plot of the chloride concentrations in the influent to the Monroe Street detention pond in Madison, Wisconsin. Chloride levels are negligible in the non-winter months but increase dramatically when road salting begins in the fall, and remain high through the snow melting period, even extending another month or so after the snowpack in the area has melted. Bennett et al. (1981) found that suspended solids and COD loadings for snowmelt runoff were about one-half of those for rainfall. Nutrients were much lower for snowmelt, while the loadings for lead were about the same for both forms of precipitation. Oberts (1994) reports that much of the annual pollutant yields from event flows in Minneapolis is accounted for by end-of-winter major melts. End-of-winter melts yielded 8 to 20 percent of the total phosphorous and total lead annual load in Minnesota. Small midwinter melts accounted for less than 5 percent of the total loads. Box 3-8 shows mass pollutant discharges for a study site in Toronto and emphasizes the significance of snowmelt discharges on the total annual storm drainage discharges.

![Chloride Concentration in the Inlet Water of the Monroe Street Detention Pond](image_url)

The Contribution of Dry Weather Discharges and Snowmelt to Overall Runoff in Toronto, Ontario

An extensive analysis of all types of stormwater flow—for both dry and wet weather—was conducted in Toronto in the mid-1980s (Pitt and McLean, 1986). The Toronto Area Watershed Management Strategy study included comprehensive monitoring in a residential/commercial area and an industrial area for summer stormwater, warm season dry weather flows, snowmelt, and cold season dry weather flows. In addition to the outfall monitoring, detailed source area sheet flow monitoring was also conducted during rain and snowmelt events to determine the relative magnitude of pollutant sources. Particulate accumulation and wash-off tests were also conducted for a variety of streets in order to better determine their role in contaminant contributions.

Tables 3-9 and 3-10 summarize Toronto residential/commercial and industrial urban runoff median concentrations during both warm and cold weather, respectively. These tables show the relative volumes and concentrations of wet weather and dry weather flows coming from the different land uses. The bacteria densities during cold weather are substantially less than during warm weather, but are still relatively high; similar findings were noted during the NURP studies (EPA, 1983). However, chloride concentrations and dissolved solids are much higher during cold weather. Early spring stormwater events also contain high dissolved solids concentrations. Cold weather runoff accounted for more than half of the heavy metal discharges in the residential/commercial area, while warm weather discharges of zinc were much greater than the cold weather discharges for the industrial area. Warm weather flows were also the predominant sources of phosphorus for the industrial area.

One of the interesting observations is that, at these monitoring locations, warm weather stormwater runoff only contributed about 20 to 30 percent of the total annual flows being discharged from the separate stormwater outfalls. The magnitudes of the base flows were especially surprising, as these monitoring locations were research sites to investigate stormwater processes and were carefully investigated to ensure that they did not have significant inappropriate discharges before they were selected for the monitoring programs.

In comparing runoff from the industrial and residential catchments, Pitt and McLean (1986) observed that concentrations of most constituents in runoff from the industrial watershed were typically greater than the concentrations of the same constituents in the residential runoff. The only constituents with a unit-area yield that were lower in the industrial area were chlorides and total dissolved solids, which was attributed to the use of road deicing salts in residential areas. Annual yields of several constituents (total solids, total dissolved solids, chlorides, ammonia nitrogen, and phenolics) were dominated by cold weather flows, irrespective of the land use.

A comparison of the Toronto sheet flow data from the different land-use areas indicated that the highest concentrations of lead and zinc were found in samples collected from paved areas and roads during both rain runoff and snowmelt (Pitt and McLean, 1986). Fecal coliform values were significantly higher on sidewalks and on, or near, roads during snowmelt sampling, likely because these areas are where dogs would be walked in winter conditions. In warm weather, dog walking would be less concentrated into these areas.

The concentrations for total solids from grass or bare open areas were reduced dramatically during snowmelt compared to rain runoff, an indication of the reduced erosion and the...
poor delivery of particulate pollutants during snowmelt periods. Cold weather sheet flow median concentrations of particulate solids for the grass and open areas (80 mg/L) were much less than the TSS concentrations observed during warm weather runoff (250 mg/L) for these same areas. Snowmelt total solids concentrations also increased in areas located near roads due to the influence of road salting on dissolved solids concentrations. In the residential areas, streets were the most significant source of snowmelt solids, while yards and open areas were the major sources of nutrients. Parking and storage areas contrib-

TABLE 3-9 Median Pollutant Concentrations Observed at Toronto Outfalls during Warm Weather

<table>
<thead>
<tr>
<th>Measured Parameter</th>
<th>Baseflow</th>
<th>Stormwater</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Residential</td>
<td>Industrial</td>
</tr>
<tr>
<td>Stormwater volume (m$^3$/ha/season)</td>
<td>950</td>
<td>1500</td>
</tr>
<tr>
<td>Baseflow volume (m$^3$/ha/season)</td>
<td>1700</td>
<td>2100</td>
</tr>
<tr>
<td>Total residue</td>
<td>256</td>
<td>371</td>
</tr>
<tr>
<td>Total dissolved solids</td>
<td>973</td>
<td>454</td>
</tr>
<tr>
<td>Suspended solids</td>
<td>43</td>
<td>22</td>
</tr>
<tr>
<td>Chlorides</td>
<td>78</td>
<td>34</td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>0.09</td>
<td>0.28</td>
</tr>
<tr>
<td>Phosphates</td>
<td>0.12</td>
<td>0.02</td>
</tr>
<tr>
<td>Total Kjeldahl nitrogen (organic N plus NH$_3$)</td>
<td>0.9</td>
<td>2.4</td>
</tr>
<tr>
<td>Ammonia nitrogen</td>
<td>&lt;0.1</td>
<td>&lt;0.1</td>
</tr>
<tr>
<td>Chemical oxygen demand</td>
<td>22</td>
<td>108</td>
</tr>
<tr>
<td>Fecal coliform bacteria (#/100 mL)</td>
<td>33,000</td>
<td>7,000</td>
</tr>
<tr>
<td>Fecal strep. bacteria (#/100 mL)</td>
<td>2,300</td>
<td>8,800</td>
</tr>
<tr>
<td>Pseudomonas aeruginosa bacteria (#/100 mL)</td>
<td>2,900</td>
<td>2,380</td>
</tr>
<tr>
<td>Cadmium</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Chromium</td>
<td>&lt;0.04</td>
<td>0.42</td>
</tr>
<tr>
<td>Copper</td>
<td>0.02</td>
<td>0.05</td>
</tr>
<tr>
<td>Lead</td>
<td>&lt;0.04</td>
<td>&lt;0.04</td>
</tr>
<tr>
<td>Zinc</td>
<td>0.04</td>
<td>0.18</td>
</tr>
<tr>
<td>Phenolics (µg/L)</td>
<td>&lt;1.5</td>
<td>2.0</td>
</tr>
<tr>
<td>α-BHC (ng/L)</td>
<td>17</td>
<td>&lt;1</td>
</tr>
<tr>
<td>γ-BHC (lindane) (ng/L)</td>
<td>5</td>
<td>&lt;2</td>
</tr>
<tr>
<td>Chlordane (ng/L)</td>
<td>4</td>
<td>&lt;2</td>
</tr>
<tr>
<td>Dieldrin (ng/L)</td>
<td>4</td>
<td>&lt;5</td>
</tr>
<tr>
<td>Pentachlorophenol (ng/L)</td>
<td>280</td>
<td>50</td>
</tr>
</tbody>
</table>

Values are in mg/L unless otherwise indicated. Warm weather samples were obtained during the late spring, summer, and early fall months when the air temperatures were above freezing and no snow was present.

SOURCE: Pitt and McLean (1986).
uted the most snowmelt pollutants in the industrial area. An analysis of snow samples taken along a transect of a snowpack adjacent to an industrial road showed that the pollut-
ant levels decreased as a function of distance from the roadway. At distances greater than 3 to 5 meters from the edge of the snowpack, the concentrations were relatively constant. Novotny et al. (1986) sampled along a transect of a snowpack by a freeway in Milwaukee. They also found that the concentration of constituents decreased as the distance from the road increased. Most of the measured constituents, including total solids and lead, were at or near background levels at 30 meters or more from the road.

### TABLE 3-10 Median Pollutant Concentrations Observed at Toronto Outfalls during Cold Weather

<table>
<thead>
<tr>
<th>Measured Parameter</th>
<th>Baseflow</th>
<th>Snowmelt</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stormwater volume (m³/ha/season)</td>
<td>—</td>
<td>1800</td>
</tr>
<tr>
<td>— (Industrial)</td>
<td>830</td>
<td></td>
</tr>
<tr>
<td>Base flow volume (m³/ha/season)</td>
<td>1100</td>
<td>660</td>
</tr>
<tr>
<td>Total residue</td>
<td>2230</td>
<td>1080</td>
</tr>
<tr>
<td>Total dissolved solids</td>
<td>2210</td>
<td>1020</td>
</tr>
<tr>
<td>Suspended solids</td>
<td>21</td>
<td>50</td>
</tr>
<tr>
<td>Chlorides</td>
<td>1080</td>
<td>470</td>
</tr>
<tr>
<td>Total phosphorus</td>
<td>0.18</td>
<td>0.34</td>
</tr>
<tr>
<td>Phosphates</td>
<td>&lt;0.05</td>
<td>&lt;0.02</td>
</tr>
<tr>
<td>Total Kjeldahl nitrogen (organic N plus NH₃)</td>
<td>1.4</td>
<td>2.0</td>
</tr>
<tr>
<td>Ammonia nitrogen</td>
<td>&lt;0.1</td>
<td>&lt;0.1</td>
</tr>
<tr>
<td>Chemical oxygen demand</td>
<td>48</td>
<td>68</td>
</tr>
<tr>
<td>Fecal coliform bacteria (#/100 mL)</td>
<td>9800</td>
<td>400</td>
</tr>
<tr>
<td>Fecal strep bacteria (#/100 mL)</td>
<td>1400</td>
<td>2400</td>
</tr>
<tr>
<td>Pseudomonas aeruginosa bacteria (#/100 mL)</td>
<td>85</td>
<td>55</td>
</tr>
<tr>
<td>Cadmium</td>
<td>&lt;0.01</td>
<td>&lt;0.01</td>
</tr>
<tr>
<td>Chromium</td>
<td>&lt;0.01</td>
<td>0.24</td>
</tr>
<tr>
<td>Copper</td>
<td>0.02</td>
<td>0.04</td>
</tr>
<tr>
<td>Lead</td>
<td>&lt;0.06</td>
<td>&lt;0.04</td>
</tr>
<tr>
<td>Zinc</td>
<td>0.07</td>
<td>0.15</td>
</tr>
<tr>
<td>Phenolics (mg/L)</td>
<td>2.0</td>
<td>7.3</td>
</tr>
<tr>
<td>α-BHC (ng/L)</td>
<td>NA</td>
<td>3</td>
</tr>
<tr>
<td>γ-BHC (lindane) (ng/L)</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Chlordane (ng/L)</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Dieldrin (ng/L)</td>
<td>NA</td>
<td>NA</td>
</tr>
<tr>
<td>Pentachlorophenol (ng/L)</td>
<td>NA</td>
<td>NA</td>
</tr>
</tbody>
</table>

1Values are in mg/L unless otherwise indicated. Cold weather samples were obtained during the winter months when the air temperatures were commonly below freezing. Snowmelt samples were obtained during snowmelt episodes and when rain fell on snow.
NA, not analyzed
SOURCE: Pitt and McLean (1986).
Atmospheric Deposition

The atmosphere contains a diverse array of contaminants, including metals (e.g., copper, chromium, lead, mercury, zinc), nutrients (nitrogen, phosphorus), and organic compounds (e.g., PAHs, polychlorinated biphenyls, pesticides). These contaminants are introduced to the atmosphere by a variety of sources, including local point sources (e.g., power plant stacks) and mobile sources (e.g., motor vehicles), local fugitive emissions (e.g., street dust and wind-eroded materials), and transport from non-local areas. These emissions, composed of gases, small particles (aerosols), and larger particles, become entrained in the atmosphere and subject to a complex series of physical and chemical reactions (Schueler, 1983).

Atmospheric contaminants are deposited on land and water in two ways—termed wet deposition and dry deposition. Wet deposition (or wetfall) involves the sorption and condensation of pollutants to water drops and snowflakes followed by deposition with precipitation. This mechanism dominates the deposition of gases and aerosol particles. Dry deposition (or dryfall) is the direct transfer of contaminants to land or water by gravity (particles) or by diffusion (vapor and particles). Dry deposition occurs when atmospheric turbulence is not sufficient to counteract the tendency of particles to fall out at a rate governed, but not exclusively determined, by gravity (Schueler, 1983).

As atmospheric contaminants deposit, they can exert an influence on stormwater in several ways. Contaminants deposited by wetfall are directly conveyed to stormwater while those in dryfall can be washed off the land surface. For both processes, the atmospheric load of contaminants is strongly influenced by characteristics such as the amount of impervious surface, the magnitude and proximity of emission sources, wind speed and direction, and precipitation magnitude and frequency (Schueler, 1983). Deposition rates can depend on the type of contaminant and can be site-specific. The relationships between atmospheric deposition and stormwater quality are, however, not well understood and difficult to determine. Following are a few illustrative examples.

Southern California

Several studies have addressed atmospheric deposition in Southern California (e.g., Lu et al., 2003; Harris and Davidson, 2005; Stolzenbach et al., 2007). Stolzenbach et al. and Lu et al. conclude the following for this region:

- the major source of contaminants to the atmosphere in this region is associated with resuspended dust, primarily from roads,
- contaminants in resuspended dust may reflect historical as well as current sources and distant as well as local sources,
- atmospheric loadings to the receiving water are primarily the result of chronic daily dry deposition of large particles greater than 10 µm in size on the
watershed rather than directly on a waterbody,
  • significant spatial variability occurs in trace metal mass loadings and
deposition fluxes, particularly along transportation corridors along the coast and
the mountain slopes of the airshed,
  • significant diurnal and seasonal variations occur in the deposition of
trace metals, and
  • atmospheric deposition of metals is a significant component of con-
taminant loading to waterbodies in the region relative to other point and non-
point sources.

Harris and Davidson (2005) have reported that traditional sources of lead to
the south coast air basin of California accounted for less than 15 percent of the
lead exiting the basin each year. They resolve this difference by considering
that lead particles deposited during the years of leaded gasoline use are resus-
pended as airborne lead at this time, some decades after their original deposition.
This result indicates that lead levels in the soil will remain elevated for decades
and that resuspension of this lead will remain a major source of atmospheric
lead well into the future.

Sabin et al. (2005) assessed the contribution of trace metals (chromium,
copper, lead, nickel, and zinc) from atmospheric deposition to stormwater runoff
in a small impervious urban catchment in the Los Angeles area. Dry deposition
contributed 90 percent or more of the total deposition inside the catchment, indi-
cating the dominance of dry deposition in semi-arid regions such as Los Ange-
les. Deposition potentially accounted for from 57 to 90 percent of the total trace
metals in stormwater in the study area, demonstrating that atmospheric deposi-
tion can be an important source of trace metals in stormwater near urban centers.

San Francisco

Dissolved copper is toxic to phytoplankton, the base of the aquatic food
chain. Copper and other metals are released in small quantities when drivers
depress their brakes. The Brake Pad Partnership (http://www.suscon.org/
brakepad/index/asp) has conducted studies to determine how much copper is
released as wear debris, and how it travels through the air and streets to surface
waters. A comprehensive and complex model of copper loads to and of trans-
port and reactions in San Francisco Bay was developed (Yee and Franz, 2005).
Objectives were to provide daily loadings of flow, TSS, and copper to the bay
and to estimate the relative contribution of brake pad wear debris to copper in
the bay. The modeling results (Rosselot, 2006a) indicated that an estimated
47,000 kg of copper was released to the atmosphere in the Bay Area in 2003. Of
this amount, 17,000 kg Cu/yr was dry-deposited in subwatersheds; 3,200 kg
Cu/yr was wet-deposited in subwatersheds; 1,200 kg Cu/yr was dry-deposited
directly to bay waters; and 1,300 kg Cu/yr was wet-deposited directly to bay
waters. The remaining 24,000 kg Cu/yr remained airborne until it left the Bay

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Area. The contribution of copper from brake pads to the bay is estimated to range from 10 to 35 percent of the total copper input, with the best estimate being 23 percent (Rosselot, 2006a,b).

**Washington, D.C., Metropolitan Area**

Schueler (1983) investigated the atmospheric deposition of several contaminants in Washington, D.C., and its surrounding areas in the early 1980s. The contaminants assessed included trace metals (cadmium, copper, iron, lead, nickel, and zinc), nutrients (nitrogen and phosphorus), solids, and organics as measured collectively by BOD and COD. Dryfall solids loading increased progressively from rural to urban sites. A similar trend was observed for total phosphorus, total nitrogen, and trace metal dry deposition rates. Wet deposition rates exhibited few consistent regional patterns.

The relative importance of wet and dry deposition varied considerably with each contaminant and each site. For example, most of the nitrogen was supplied by wet deposition while most of the phosphorus was delivered via dry deposition. If a contaminant is deposited primarily by wet deposition, it is likely that a major fraction of it will be rapidly entrained in urban runoff.

Atmospheric sources were estimated to contribute from 70 to 95 percent of the total nitrogen load to urban runoff and 20 to 35 percent of the total phosphorus load. Overall, atmospheric deposition appeared to be a moderate source of pollutants in urban runoff. However, with the exception of nitrogen, atmospheric deposition was not the major source.

Average annual atmospheric deposition rates suggested a general trend toward greater deposition rates from rural to suburban to urban sites. This pattern was most pronounced for dry deposition. Wet deposition was the most important deposition mechanism for total nitrogen, nitrate, organic nitrogen, COD, copper, and zinc. Dry deposition was most important for most soil-related constituents, such as total solids, iron, lead, total phosphorus, and orthophosphate.

Measurements of rainfall pH showed median values between 4.0 and 4.1 at all stations and during all seasons. Increased mobilization of trace metals from urban surfaces caused by acid rain was noted at several monitoring sites.

***

Relationships between atmospheric deposition rates and the quality of urban stormwater are complex and cannot be generalized regionally or temporally. Site-specific measurements or reliable estimates of (1) contaminant sources, (2) atmospheric particle size and contaminant concentrations, (3) deposition rates and mechanisms, (4) land surface characteristics, (5) local and regional hydrology and meteorology, and (6) contaminant concentrations in stormwater are needed to assess management decisions to improve stormwater quality. Transportation is a major source of metals (lead in gasoline, zinc in tires, copper in...
EFFECTS OF URBANIZATION ON WATERSHEDS 207

brake pads). The results of the modeling of copper in San Francisco and its watershed demonstrate the feasibility of modeling the impact of a source, in this case copper input by atmospheric deposition, on water quality in a receiving waterbody.

**BIOLOGICAL RESPONSES TO URBANIZATION**

As discussed in Chapter 1, the biological integrity of aquatic ecosystems is influenced by five major categories of environmental stressors: (1) chemical, (2) hydrologic, (3) physical (e.g., habitat), (4) biological (e.g., disease, alien species), and (5) energy-related factors (e.g., nutrient dynamics). Recent studies on biological assemblages in urban or urbanizing waters have begun to examine how stormwater stressors limit biological potential along various urban gradients (Horner et al., 2003; Carter and Fend, 2005; Meador et al., 2005; Barbour et al., 2008; Purcell et al., 2009). Advances in biological monitoring and assessment over the past two decades have enabled much of this research. Today, many states and tribes use biological data to directly measure their aquatic life beneficial uses and have developed numeric biocriteria that are institutionalized in their water quality standards. Most of these approaches compare biology and stressors to suites of reference sites (Hughes, 1995; Stoddard et al., 2006), which can vary from near-pristine areas to agricultural landscapes. While this section focuses on streams because of the wealth of data, similar work is being performed on other waterbody types such as wetlands (Mack and Micacchion, 2007) and estuaries, both of which are susceptible to stormwater pollutants such as metals because of their depositional nature (Morrisey et al., 2000).

Aquatic life beneficial uses are based on achieving aquatic potential given feasible restorative actions. Because such potential may vary substantially across a region depending on land use and other factors, some states have adopted tiered aquatic life uses (see Box 2-1). The potential of many urban streams is likely to be something less than “biological integrity” (the ultimate goal of the CWA) or even “fishable–swimmable” goals, which are the interim goals of the CWA. Indeed, there is a near-universal, negative association between biological assemblages in streams and increasing urbanization, to the extent that it has been termed the “Urban Stream Syndrome” (Walsh et al., 2005). Recent investigations that have quantified the responses of macroinvertebrates and other biological assemblages along multiple measures of urban/stormwater stressors have discussed how best to set aquatic life goals for urban streams (Booth and Jackson, 1997; Bernhardt and Palmer, 2007). One of the most important contributions to this debate has been the development of the Biological Condition Gradient (BCG) concept by EPA. The BCG is an attempt to anchor and standardize interpretations of biological conditions and to unify biological monitoring results across the United States in order to advance the use of tiered aquatic life beneficial uses. This section summarizes the characteristic biological responses to urban gradients, within the framework of the BCG, and it re-
views evidence of biological responses within the aforementioned five major
categories of environmental stressors.

**Biological Condition Gradient**

The BCG framework is an ecological model of how structural and functional components of biological assemblages change along gradients of increasing stressors of many kinds (Davies and Jackson, 2006). Ecological systems have some common general attributes related to their structure and function that form the basis for how biological organisms respond to stressors in the environment. Over the past 20 years, development of biological indicators nationwide has taken advantage of these repeatable biological responses to stress; however, state benchmarks often have varied substantially, even between adjacent states. To gain consistency, the EPA convened a national workgroup of EPA Regions, States, and Tribes to develop the BCG—a standardized, nationally applicable model that defines important attributes of biological assemblages and describes how these attributes change along a gradient of increasing stress from pristine environments to severely impaired conditions (Figure 3-38; Davies and Jackson, 2006). The goals of this work were to improve national consistency in the rating and application of biological assessment tools for all types of waterbodies and to provide a baseline for the development of tiered aquatic life used.

To date, the BCG has been applied to assemblages including aquatic macroinvertebrates, fish, Unionid mussels, and algae in streams, but it could be applied to any organism group in any type of waterbody. The BCG is derived by applying a suite of ten ecological attributes that allows biological condition to be interpreted independently of assessment method (Table 3-11; Davies and Jackson, 2006). The first five attributes focus on taxa sensitivity, an important component of tools such as multimetric indices (e.g., the Index of Biotic Integrity [IBI], the Invertebrate Community Index [ICI]; see Box 2-3) used in the United States and Europe. Many indicator taxa have been widely studied, and, for groups such as fish, historical data often exist. Most states have established lists of tolerant and intolerant species as part of their use of biological indices (Simon and Lyons, 1995). The relatively large literature on species population and distribution changes in response to stressors and landscape condition offers insight into the mechanisms for population shifts, some of which are summarized in this section.

The first two attributes of the BCG relate to those streams that are closest to natural or pristine, with most taxa “as naturally occur.” Attribute 1 and 2 taxa are the most sensitive species that typically disappear with even minor stress. Table 3-12 lists some example attribute 1 taxa for four different regions of the United States. Attribute 3 reflects more ubiquitous, but still sensitive, species that can provide information as human influence on the landscape becomes more obvious, but is not yet severe. Attributes 5 and 6 are taxa that increase in abundance and distribution with increasing stress. The organism condition at-
The Biological Condition Gradient: Biological Response to Increasing Levels of Stress

<table>
<thead>
<tr>
<th>Levels of Biological Condition</th>
<th>Biological Condition</th>
<th>Level of Exposure to Stressors</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural structural, functional, and taxonomic integrity is preserved.</td>
<td>Natural structural, functional, and taxonomic integrity is preserved.</td>
<td>Watershed, habitat, flow regime and water chemistry as naturally occurs.</td>
</tr>
<tr>
<td>Structure &amp; function similar to natural community with some additional taxa &amp; biomass; ecosystem level functions are fully maintained.</td>
<td>Structure &amp; function similar to natural community with some additional taxa &amp; biomass; ecosystem level functions are fully maintained.</td>
<td>Chemistry, habitat, and/or flow regime severely altered from natural conditions.</td>
</tr>
<tr>
<td>Evident changes in structure due to loss of some rare native taxa, shift in relative abundance of native taxa, loss of some rare native taxa; ecosystem functions are largely maintained.</td>
<td>Evident changes in structure due to loss of some rare native taxa, shift in relative abundance of native taxa, loss of some rare native taxa; ecosystem functions are largely maintained.</td>
<td></td>
</tr>
<tr>
<td>Moderate changes in structure due to replacement of sensitive ubiquitous taxa by more tolerant taxa, ecosystem functions largely maintained.</td>
<td>Moderate changes in structure due to replacement of sensitive ubiquitous taxa by more tolerant taxa, ecosystem functions largely maintained.</td>
<td></td>
</tr>
<tr>
<td>Sensitive taxa markedly diminished; conspicuously unbalanced distribution of major taxonomic groups; ecosystem function shows reduced complexity &amp; redundancy.</td>
<td>Sensitive taxa markedly diminished; conspicuously unbalanced distribution of major taxonomic groups; ecosystem function shows reduced complexity &amp; redundancy.</td>
<td></td>
</tr>
<tr>
<td>Extreme changes in structure and ecosystem function; wholesale changes in taxonomic composition; extreme alterations from normal densities.</td>
<td>Extreme changes in structure and ecosystem function; wholesale changes in taxonomic composition; extreme alterations from normal densities.</td>
<td></td>
</tr>
</tbody>
</table>

FIGURE 3-38 The Biological Condition Gradient (BCG) and summaries of biological condition along tiers of this gradient. SOURCE: Modified from Davies and Jackson (2006) by EPA.

TABLE 3-11 Ecological attributes that comprise the basis for the BCG

<table>
<thead>
<tr>
<th>Ecological Attributes</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Historically documented, sensitive, long-lived or regionally endemic taxa</td>
</tr>
<tr>
<td>2. Sensitive-rare taxa</td>
</tr>
<tr>
<td>3. Sensitive-ubiquitous taxa</td>
</tr>
<tr>
<td>4. Taxa of intermediate tolerance</td>
</tr>
<tr>
<td>5. Tolerant taxa</td>
</tr>
<tr>
<td>6. Non-native or introduced taxa</td>
</tr>
<tr>
<td>7. Organism condition</td>
</tr>
<tr>
<td>8. Ecosystem functions</td>
</tr>
<tr>
<td>9. Spatial and temporal extent of detrimental effects</td>
</tr>
<tr>
<td>10. Ecosystem connectance</td>
</tr>
</tbody>
</table>

TABLE 3-12 Example of Taxa that Might Serve as Attribute 1: “Historically Documented, Sensitive, Long-Lived, Regionally Endemic Taxa for Streams in Four Regions of the United States”

<table>
<thead>
<tr>
<th>State and Taxon</th>
<th>Taxa Representative of Attribute 1</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maine</td>
<td></td>
</tr>
<tr>
<td>Mollusks</td>
<td>brook floater (Alasmodonta varicosa), triangle floater (Alasmodonta undulata), yellow lampmussel (Lampsilis cariosa)</td>
</tr>
<tr>
<td>Fishes</td>
<td>brook stickleback (Culaea inconstans), swamp darter (Etheostoma fusiforme)</td>
</tr>
<tr>
<td>Washington</td>
<td></td>
</tr>
<tr>
<td>Fishes</td>
<td>steelhead (Oncorhynchus mykiss)</td>
</tr>
<tr>
<td>Amphibians</td>
<td>spotted frog (Rana pretiosa)</td>
</tr>
<tr>
<td>Arizona</td>
<td></td>
</tr>
<tr>
<td>Mollusks</td>
<td>spring snails (Pyrgulopsis spp.)</td>
</tr>
<tr>
<td>Fishes</td>
<td>Gila trout (Oncorhynchus gilaee), Apache trout (Oncorhynchus apache), cutthroat trout (endemic strains) (Oncorhynchus clarkii)</td>
</tr>
<tr>
<td>Amphibians</td>
<td>Chihuahua leopard frog (Rana chiriachuensis)</td>
</tr>
<tr>
<td>Kansas</td>
<td></td>
</tr>
<tr>
<td>Mollusks†</td>
<td>hickorynut (Obovaria olivaria), black sandshell (Ligumia recta), ponderous campeloma (Campeloma crassulum)</td>
</tr>
<tr>
<td>Fishes</td>
<td>Arkansas River shiner (Notropis girardi), Topeka shiner (Notropis topeka), Arkansas darter (Etheostoma cragini), Neosho madtom (Noturus placidus), flathead chub (Platygobio gracilis)</td>
</tr>
<tr>
<td>Other</td>
<td>ringed crayfish (Orconectes neglectus neglectus), Plains sand-burrowing mayfly (Homoeoneuria ammaphila)</td>
</tr>
<tr>
<td>invertebrates</td>
<td></td>
</tr>
<tr>
<td>Amphibians</td>
<td>Plains spadefoot toad (Spea bombifrans), Great Plains toad (Bubo cognatus), Great Plains narrowmouth toad (Gastrophyne olivaceae), Plains leopard frog (Rana blairi)</td>
</tr>
</tbody>
</table>

†Although not truly endemic to the central plains, these regionally extirpated mollusks were widely distributed in eastern Kansas prior to the onset of intensive agriculture.


tribute (7) includes the presence of anomalies (e.g., tumors, lesions, eroded fins, etc.) or the presence of large or long-lived individuals in a population. Most natural streams typically have few or incidental rates of “anomalies” associated with disease and stress. Natural waterbodies typically also have the entire range of life stages present, as would be expected. However, as stress is increased, larger individuals may disappear or emigrate, or reproductive failure may occur. Ecosystem function (attribute 8) is very difficult to measure directly (Davies and Jackson, 2006). However, certain functions can be inferred from structural measures common to various multimetric indices, examples of which are listed in Table 3-13. The last two attributes (9 and 10) may be of particular importance with regard to stormwater and urban impacts. Cumulative impacts are a characteristic of urbanization, and biological organisms typically integrate the effects of many small insults to the landscape. Additionally, most natural systems often have strong “connectance,” such that aquatic life often has stages that rely on migrating across multiple types or sizes of waterbodies. Urbanized streams can decrease connectance by creating migration blocks, including vertical barriers at road crossings and small dams (Warren and Pardew, 1998).
TABLE 3-13  Function Ecological Attributes or Process Rates and Their Structural Indicators

<table>
<thead>
<tr>
<th>Biotic Level and Function or Process</th>
<th>Structural Indicator</th>
</tr>
</thead>
<tbody>
<tr>
<td>Individual level</td>
<td></td>
</tr>
<tr>
<td>Fecundity</td>
<td>Maximum individual size, number of eggs</td>
</tr>
<tr>
<td>Growth and metabolism</td>
<td>Length/mass (condition)</td>
</tr>
<tr>
<td>Morbidity</td>
<td>Percentage anomalies</td>
</tr>
<tr>
<td>Population Level</td>
<td></td>
</tr>
<tr>
<td>Growth and fecundity</td>
<td>Density</td>
</tr>
<tr>
<td>Mortality</td>
<td>Size- or age-class distribution</td>
</tr>
<tr>
<td>Production</td>
<td>Biomass, standing crop, catch per unit effort</td>
</tr>
<tr>
<td>Sustainability</td>
<td>Size- or age-class distribution</td>
</tr>
<tr>
<td>Migration, reproduction</td>
<td>Presence or absence, density</td>
</tr>
<tr>
<td>Community or assemblage level</td>
<td></td>
</tr>
<tr>
<td>Production/respiration ratio,</td>
<td>Trophic guilds, indicator species</td>
</tr>
<tr>
<td>autotrophy vs heterotrophy</td>
<td></td>
</tr>
<tr>
<td>Primary production</td>
<td>Biomass, ash-free dry mass</td>
</tr>
<tr>
<td>Ecosystem level</td>
<td></td>
</tr>
<tr>
<td>Connectivity</td>
<td>Degree of aquatic and riparian fragmentation</td>
</tr>
<tr>
<td></td>
<td>Longitudinally, vertically, and horizontally; presence</td>
</tr>
<tr>
<td></td>
<td>or absence of diadromous and potadromous species</td>
</tr>
</tbody>
</table>


Construction of a BCG creates a conceptual framework for developing stressor–response gradients for particular urban areas. The initial work done to develop the BCG derived a series of six tiers to describe a gradient of biological condition that is anchored in pristine conditions (“as naturally occurs”) and that extends to severely degraded conditions (see Figure 3-38). Exercises done by the national work group to derive such a gradient for macroinvertebrates in wadeable streams showed strong consistency in assigning tiers to datasets using the descriptions of taxa for each attribute along these gradients (Davies and Jackson, 2006). Substantial data already exist to populate many of the attributes of the BCG and to provide mechanistic underpinning for the expected directions of change.

The BCG is not a replacement for assessment tools such as the IBI or multivariate predictive models (e.g., RIVPACS approach), but rather a conceptual overlay for characterizing the anchor point-of-reference conditions and a consistent way to communicate biological condition along gradients of stress. As such, it has strong application to understanding stormwater impacts and to communicating where a goal is located along the gradient of biological condition. While most urban goals may be distant from “pristine” or “natural,” the BCG process can dispel misconceptions that alternate urban goals are “dead streams” or unsafe in some manner.
Factors Limiting Aquatic Assemblages in Urban Waters

A slew of recent investigations have quantified the responses of macroinvertebrates and other biological assemblages to multiple measures of urbanization and to stormwater in particular. One important conclusion of some of this work is that declines in the highest biological condition start with low levels of anthropogenic change (e.g., 5 to 25 percent impervious surface); higher levels of urbanization severely alter aquatic conditions (Horner et al., 2003). This has important consequences for protecting sites with the highest biological integrity, as they may be among the most vulnerable. The non-threshold nature of this aquatic response and the typical wedge-shaped response to multiple stressors by aquatic assemblages are discussed in Box 3-9.

BOX 3-9
Non-threshold Nature of the Decline of Biological Assemblages Along Urban Stressor Gradients

Several recent surveys have demonstrated that biological assemblages begin to decline in condition with even low levels of urban disturbance as measured by various gradients of urbanization (e.g., May, 1996; Horner et al., 1997; May et al., 1997; Horner et al., 2003; Moore and Palmer, 2005; Barbour et al., 2008). This box summarizes the work of Horner et al. (2003) in small streams in three regions: Montgomery County, Maryland; Austin, Texas; and the Puget Sound area of Washington. Geographic Information System (GIS) analyses using information such as land use, total impervious area, and riparian land use were used to develop multi-metric Watershed Condition Indices (WCIs) for each region. These in turn were related to fish and macroinvertebrate indices, e.g., benthic IBIs, (B-IBI, all three regions), a fish IBI (F-IBI for Maryland) and an index that was the ratio of the sensitive coho salmon to the more tolerant cutthroat trout in collections for the Puget Sound lowland area.

In each of these areas, no or extremely low urban development, substantial forest cover, and minimal disturbance of riparian zones characterized sites with the highest biological scores, but these conditions did not guarantee high scores because other impacts could limit biology even with these "natural" characteristics. In all three regions, high urbanization and loss of natural cover always led to biological degradation (Figures 3-39 and 3-40). The results of this study were similar to other recent studies such as Barbour et al. (2008) that identify a "wedge-shaped" relationship or a "polygonal" relationship (Carter and Fend, 2005) between urban gradients and biological condition. These types of relationships have also been termed "factor-ceiling" relationships (Thomson et al., 1996). The outer surface of these wedges or polygons reflects where the urban gradients limit biological assemblages, such that points below this surface typically represent sites affected by other stressors (e.g., combined sewer overflows, discharges, etc.). In all of these studies it is easier to predict loss of biological conditions as the urban gradients (e.g., WCI) worsen than it is to ensure high biological integrity at low proportions of urban stress (because some other stressor may still limit aquatic condition).

continues next page
FIGURE 3-39  Plots of a measure of urbanization (TIA + Wetland & Forest Cover + IRI) versus B-IIBIs for Austin, Texas (top), and Montgomery County, Maryland (bottom). SOURCE: Horner et al. (2003).
FIGURE 3-40 Plots of a measure of urbanization (TIA + Wetland & Forest Cover + IRI) versus B-IBIs for Puget Sound (top) and versus the ratio of coho salmon to cutthroat trout for Puget Sound (bottom). SOURCE: Horner et al. (2003).
BOX 3-9 Continued

Horner et al. (2003) also focused on whether structural SCMs could moderate the effects of urbanization on biological assemblages. They made detailed observations of two subbasins in the Puget Sound lowland area, one with a greater degree of stormwater management than the other (although neither had what would be considered comprehensive stormwater management with a focus on water quality issues). As shown in Figure 3-41, at the highest levels of urbanization (triangles), the subbasin with the more extensive use of structural SCMs did have better biological conditions. There was less evidence of biological benefit in the watershed that used SCMs but it had only moderate urbanization and more natural land cover (squares and diamonds). There were no circumstances where high biological condition was observed along with the use of SCMs because high biological condition only occurred where little human alteration was present, and thus SCMs were not used.

FIGURE 3-41 Macroinvertebrate community index versus structural SCM density with the highest, intermediate, and lowest one-third of natural watershed and riparian cover. The upper and lower horizontal lines represent indices considered to define relatively high and low levels of biological integrity, respectively. SOURCE: Horner et al. (2003).

The sections that follow review the evidence underlying biological responses to each of the major categories of stressors: chemical, hydrologic, physical habitat, biological, and energy-related factors. As will be evident in some of the examples, the stressors themselves can interact (e.g., flow can influence habitat, habitat can influence energy processing, etc.), which increases the complexity of understanding how stormwater affects aquatic ecosystems.
Biological Responses to Toxic Pollutants

The chemical constituents of natural streams vary widely with climatic region, stream size, soil types, and geological setting. Most small natural streams, outside of unique areas with naturally occurring toxicants, have very low levels of chemicals considered to be toxicants and have relatively low levels of dissolved and particulate materials in general. This applies to chemicals in the water column and in sediments. Increasing amounts of impervious surface in the watershed typically increase the concentrations of many chemical parameters in runoff derived from urban surfaces (e.g., Porcella and Sorenson, 1980; Sprague et al., 2007).

Stormwater concentrations of these pollutants can be variable and sometimes extreme or “toxic” depending on the timing of flows (e.g., first flush), although concentrations at base flows may not routinely exceed water quality benchmarks (Sprague et al., 2007). Historical deposition of toxics in sediments can also be responsible for extremely high pollutant concentrations within waterbodies, even though the stormwater discharges may no longer be active. These situations have been termed “legacy pollution” and are most commonly associated with urban centers that have a history of industrial production.

Natural constituents such as dissolved materials (e.g., chlorides), particulate material (e.g., fine sediments), nutrients (e.g., phosphorus and nitrogen compounds), as well as a myriad of man-made parameters such as heavy metals and organic chemicals (e.g., hydrocarbons, pesticides and herbicides) have been documented to be increased and at times pervasive in stormwater (Heany and Huber, 1984; Paul and Meyer, 2001; Roy et al., 2003; Gilliom et al., 2006) although specific patterns of concentrations can vary with region and ecological setting (Sprague et al., 2007). Water chemistry impacts can also arise from a complex array of permitted discharges, storm sewer discharges, and combined sewer overflows that are treated to certain limits but at times fail to remove all constituents from flows, especially when associated with storm events (Paul and Meyer, 2001).

Streams in urban settings can have increases in toxicant levels compared to background concentrations. In many instances these cases have been associated with loss of aquatic species and impairment of aquatic life goals (EPA, 2002), which are usually explained in terms of typical lethal responses. The complexity of urban systems with regard to pathways, magnitude, duration, and timing of toxicity as well as possible synergistic or antagonistic effects of mixtures of pollutants argues for a broad approach to characterizing effects including not only toxicity testing, but also novel approaches and direct monitoring of biological assemblages (Burton et al., 1999). What is problematic from a traditional management perspective is that aquatic communities may decline before exceedances of water quality criteria are evident (May et al., 1997; Horner et al., 2003).

The first three BCG attributes focus on populations of species of high to very high sensitivity, most of which are uncommon or absent in waters with any
substantial level of urbanization. Multi-metric indices such as IBI, which reflect loss of these species, decline at least linearly with increasing urbanization (e.g., Miltnner et al., 2004; Meador et al., 2005; Walters et al., 2005). Although toxicity to compounds varies with species, many species of federal and state endangered and threatened aquatic species are more sensitive than “commonly” used test species (Dwyer et al., 2005), such that the loss of aquatic species when toxicant levels exceed criteria are readily explained.

The mechanisms of species population declines in response to chemical contaminants are likely complex and not just limited to direct lethality of the pollutant. Indeed, initial chemical changes may have no “toxic” effects, but rather could change competitive and trophic dynamics by changing primary production and energy dynamics in streams. For example, exposures to aromatic and chlorinated organic compounds from sediments derived from urban areas have been found to increase the susceptibility of salmonids to the bacterial pathogen *Vibrio anguillarum* (Arkoosh et al., 2001). Recent work has found that salmonids show substantial behavioral changes from olfactory degradation related to copper at concentrations as low as 2 \(\mu\)g/L, well below copper water quality criteria and above levels measured in most stormwater-affected streams (Hecht et al., 2007; Sandahl et al., 2007). Salmonid and other fish depend extensively on olfactory cues for feeding, emigration, responding to prey and predators, social and spawning interactions, and other behaviors, such that loss or diminution of such cues may have population-level effects on these species (Sandahl et al., 2007). Copper has been shown to cause olfactory effects on other species (Beyers et al., 2001) and to impair the sensory ability of the fish lateral line (Hernandez et al., 2006), which is nearly ubiquitous in fishes and important for most freshwater species in feeding, schooling, spawning, and other behaviors.

Whole effluent toxicity testing or sediment toxicity testing may misclassify the effects of runoff and effluents in urban settings (Burton et al., 1999). Short-term toxicity tests of stormwater often result in no identified toxicity. However, longer studies (e.g., 30 days) have shown increasing toxicity with time (Masterson and Bannerman, 1994; Ramcheck and Crunkilton, 1995). This suggests that the mechanism of toxicity could be through an ingestion pathway, for example, rather than gill uptake. Metals are often in high concentrations where fine sediments accumulate, and their legacy can extend past the time period of active discharge. Metal concentrations in urban stream sediments have been associated with high rates of fish and invertebrate anomalies such as tumors, lesions, and deformities (Burton, 1992; Ingersoll et al., 1997; Smith et al., 2003).

**Biological Responses to Non-Toxicant Chemicals**

Non-toxic chemical compounds that occur in stormwater such as nutrients, dissolved oxygen (DO), pH, and dissolved solids as well as physical factors such as temperature can have impacts on aquatic life. The effects of some of these
compounds (e.g., DO, pH) have been well documented from other impacts (e.g., wastewater, mining), such that nearly all states have developed water quality criteria for these parameters. For example, nutrient enrichment in stormwater runoff has been associated with declines of biological condition in streams (Miltner and Rankin, 1998). Chloride, sulfate, and other dissolved ions that are often elevated in urban areas can have effects on osmoregulation of aquatic organisms and have been associated with loss of species sensitive to dissolved materials such as mayflies (Kennedy et al., 2004). The concentrations of these compounds can vary regionally (Sprague et al., 2007) and with the degree of urbanization.

Water quality criteria for temperature were spurred by the need for thermal permits for industrial and power plant cooling water discharges. There is a very large literature on the importance of water temperature to aquatic organisms; preference, avoidance, and lethal temperature ranges have been derived for many aquatic species (e.g., Brungs and Jones, 1977; Coutant, 1977; Eaton et al., 1995). In addition, temperature is one of the key classification strata for aquatic life, in that streams are routinely classified as cold water, cool water, or warm water based on the geographic and natural settings of waters. The removal of catchment and riparian vegetation and the general increase in surface runoff from impervious, man-made, and heat-capturing surfaces has been associated with increasing water temperatures in urban waterbodies (Wang and Kanehl, 2003; Nelson and Palmer, 2007). A number of researchers have created models to predict in-stream temperatures based on urban characteristics (Krause et al., 2004; Herb et al., 2008).

Hydrologic Influences on Aquatic Life

The importance of “natural” flow regimes on aquatic life has been well documented (Poff et al., 1997; Richter et al., 1997a, 2003). As watersheds urbanize, flow regimes change from little runoff to over 40 to 90 percent of the rainfall becoming surface runoff (Roesner and Bledsoe, 2003). Flow regimes in urban streams typically are very “flashy,” with higher and more frequent peak events, compared to undisturbed systems (Poff et al., 1997; Baker et al., 2004) and well as reduced base flows and more frequent desiccation (Bernhardt and Palmer, 2007). Richter et al. (1996) proposed a series of indicators that could be used to measure hydrologic disturbance, many of which have been used in the recent studies identifying the hydrologic effects of stormwater on aquatic biota (Barbour et al., 2008). Pomeroy et al. (2008) did an extensive review of which flow characteristics appear to have the greatest influence on biological metrics and biological integrity. No single measure of flow was found to be significant in all studies; however, important attributes included flow variability and flashiness, flood frequency, flow volume, flow variability, flow timing, and flow duration.

There are a number of mechanisms that may be responsible for the influ-
EFFECTS OF URBANIZATION ON WATERSHEDS

ence of flow characteristics on aquatic assemblages. Aquatic species vary dramatically in their swimming performance and behaviors, and species are generally adapted to undisturbed flow regimes in an area. Many low- to moderate-gradient small streams in the United States, for example, have strong connections with their flood-prone areas and often possess habitat features that insulate poor swimming species from episodic natural high flows. Undercut banks, rootwads, oxbows, and backwater habitats all can act as refugia from high flows. Some aquatic species are more or less mobile within the sediments, like certain macroinvertebrates (meiofauna or hyporheos) and fish species such as sculpins and madtoms. Secondary impacts from hydrologic changes such as bank erosion and aggradation of fines can render substrates embedded and prohibit organisms, particularly the meiofauna, from moving vertically within the bottom substrates (Schmid-Araya, 2000). Substrate fining has been documented to occur with increasing urbanization, especially in the early stages of development, which can embed spawning habitats and eliminate or reduce spawning success of fish such as salmonids and minnows (Waters, 1995).

Flood flows can cause mortality in the absence of urbanization. For example, flood flows in streams under natural conditions have been documented as a cause of substantial mortality in young or larval fish such as smallmouth bass (Funk and Fleener, 1974; Lorantas and Kristine, 2004). Increased flashiness from urbanization is likely to exacerbate this effect. Thus, increases in the frequency of peak flows during spring will increase the probability of spawning failure, such that sensitive species may eventually be locally extirpated. In urban areas, culverts and other flow obstructions can create conditions that may preclude re-colonization of upstream reaches because weak-swimming fishes cannot move past flow constrictions or leap past vertical drops caused by artificial structures.

Hydrologic simplification and stream straightening that occur in urban streams, often as a result of increased peak flows or as a local management response, typically remove habitat used as temporary refuges from high flows, such as backwater areas, undercut banks, and rootwads. There is a large literature relating populations of fish and macroinvertebrates to various habitat features of streams, rivers, and wetlands. The first two attributes of the BCG identify taxa that are historically documented, sensitive, long-lived, or regionally endemic taxa or sensitive-rare taxa. Many of these taxa are endangered because of large-scale changes in flow-influenced habitats; that is, threats of extinction often center on habitat degradation that influence spawning, feeding, or other aspects of a species life history (Rieman et al., 1993). In contrast, many of the fish and macroinvertebrate taxa that compose regional lists of tolerant taxa are tolerant to habitat changes related to flow disturbance as well as chemical parameters. Understanding the life history attributes of certain species and how they may change with multiple stressors (Power, 1997) is an important tool for understanding complex responses of aquatic ecosystems to urban stressors.
Geomorphic and Habitat Influences on Aquatic Life

In natural waters, geomorphic factors and climate, modified by vegetation and land use, constrain the types of physical habitat features likely to occur in streams (Webster and D’Angelo, 1997). For example, very-low-gradient streams may have few riffles and be dominated by woody debris and bank cover, whereas higher gradient waters may have more habitat types formed by rapidly flowing waters (riffles, runs). Aquatic life in streams is influenced directly by the habitat features that are present, such as substrate types, in-stream structures, bank structure, and flow types (e.g., deep-fast vs. shallow-slow).

As discussed previously, human alteration of landscapes, encroachment on riparian areas, and direct channel modifications (e.g., channelization) that accompany urbanization have often resulted in unstable channels, with negative consequences for aquatic habitat. As urbanization has increased, channel density has declined because streams have been piped, dewatered, and straightened (Meyer and Wallace, 2001; Paul and Meyer, 2001). Changes in the magnitude, relative proportions, and timing of sediment and water delivery have resulted in loss of aquatic life and habitat via a wide range of mechanisms, including changes in channel bed materials, increased suspended sediment loads, loss of riparian habitat due to bank erosion, and changes in the variability of flow and sediment transport characteristics relative to aquatic life cycles (Roesner and Bledsoe, 2003). There are still significant gaps in knowledge about how stormwater stressors can affect stream habitat, especially as one moves from the reach scale to the watershed scale. Understanding the stage and trajectory of channel evolution is critical to understanding channel recovery and expected habitat conditions or in choosing effective restoration options (Simon et al., 2007).

Across much of the United States, stream habitats have been altered to the imperilment of aquatic species (Williams et al., 1989; Richter et al., 1997b; Strayer et al., 2004). A study of rapidly urbanizing streams in central Ohio identified the loss of highly and moderately sensitive species as a key factor the decline in the IBI in these streams (Miltner et al., 2004). These streams had historical fish collections when they were primarily influenced by agricultural land use; sampling after the onset of suburban development documented the loss of many of these species attributable to land-use changes and habitat degradation along these urban streams. Along the BCGs that have been developed for streams, most of the species in attributes 1–3 are specialists requiring very specific habitats for spawning, feeding, and refuge. Habitat alteration, either direct or indirect, creates harsh environments that tend to favor tolerant taxa, which would otherwise be in low abundance. Often these tolerant species are characterized by high reproductive potential, generalist feeding behaviors, tolerance to chemical stressors such as low DO, and pioneering strategies that allow rapid recolonization following acute stressful events.
Altered Energy Pathways in Urban Streams

The pathways of energy flow in streams are an important determinant of aquatic species distributions. In most natural temperate streams, headwaters transform and export energy from stream side vegetation and adjacent land uses into aquatic biomass. The types, amount, and timing of delivery of water, organic material, and debris have important consequences for conditions downstream (Dolloff and Webster, 2000). The energy-transforming aspect of stream ecosystems is difficult to capture directly, so most measures are surrogates, such as the trophic characteristics of assemblages and chemical and physical characteristics consistent with natural energy processes.

An increasingly urban landscape can have a complex array of effects on energy dynamics in streams (Allan, 2004). Loss of riparian areas and changes in riparian vegetation can reduce the supply and quality of coarse organic matter that forms the base of aquatic food webs in most small streams. The reduction in the amount of organic matter with riparian loss is obvious; however, changing species of vegetation (e.g., invasion or planting of exotic species) can affect the quality of organic matter and influence higher trophic levels because, for example, exotic species may have different nutrient values (e.g., C/N ratios, trace chemicals) or process nutrients at a different rate (Royer et al., 1999). Furthermore, native invertebrate taxa may not be adapted to utilize the exotic material (Miller and Boulton, 2005). For example, changes in leaf species in a stream may alter the macroinvertebrate community by favoring species that feed on fast-decaying versus slow-decaying leaves (Smock and MacGregor, 1988; Cummins et al., 1989; Gregory et al., 1991).

Other recent work is examining ways that changes in geomorphology with increasing urbanization can influence trophic structure in streams (Doyle, 2006). Groffman et al. (2005) examined nitrogen processing in stream geomorphic structures such as bars, riffles, and debris dams in suburban and forested areas. Although suburban areas had high rates of production in organic-rich debris dams and gravel bars, higher storm flow effects in urban streams may make these features less stable and able to be maintained (Groffman et al., 2005). Changes in habitat and riparian vegetation may greatly alter trophic patterns of energy transport. For example, local nutrient enrichments combined with reduced riparian vegetation can result in nuisance algal growths in waterbodies that are evidence of simpler energy pathways. Corresponding effects are further water chemistry changes from algal decomposition (e.g., low DO) or very high algal activity (e.g., high pH) (Ehlinger et al., 2004).

The complexity of energy flow through simple ecosystems is illustrated in Figure 3-42, a “simplified” food web of a headwater stream published by Meyer (1994). The forms in which nutrients are delivered to streams may be more important than actual concentrations as well as the availability of carbon sources essential for nutrient transformation. The nutrient components that form the base of the food web in Figure 3-42 are the FPOM and CPOM boxes. In many natural streams, woody and leafy debris are the most common form of nutrient...
input, and changes to urban landscapes often change this to dissolved and finer forms. Urbanization can also reduce the retention of organic debris of streams (Groffman et al., 2005) and the timing of nutrient delivery. Timing can be of crucial importance since species spawning and growth periods may be specifically timed to take advantage of available nutrients.

As important as energy and nutrient dynamics are to stream function, many of the stream characteristics that determine effective energy flow are not typically considered when characterizing stormwater impacts. The best chance for considering these variables and maximizing ecosystem function is through inte-
EFFECTS OF URBANIZATION ON WATERSHEDS

Biological Interactions in Urban Streams

Streams in urbanized environments often are characterized by fewer native and more alien species than natural streams (DeVivo, 1996; Meador et al., 2005). The influence of exotic species is not always predictable and may be most severe in lentic environments (e.g., wetlands, estuaries) and in riparian zones where various exotic aquatic plants can greatly alter natural systems in both structure and function (Hood and Naiman, 2000). Riley et al. (2005) found that the presence of alien aquatic amphibians was positively related to degree of urbanization, as was the absence of certain native amphibian species. In a review of possible reasons for this observation, he suggested that altered flow regimes were responsible. In the arid California streams they studied, flow became more constant with urbanization (i.e., natural streams were generally ephemeral), which allowed invasion by exotic species that can prey on, compete with, or hybridize with native species (Riley et al., 2005). The alteration of stream habitat that accompanies urbanization can also lead to predation by domestic cats and dogs or collection by humans, especially where species (e.g., California newts) are large and conspicuous (Riley et al., 2005).

The effects of specific exotic species on aquatic systems have been observed to vary geographically, although recent work has found correlations between total invasion rate and the number of high-impact exotic species (Ricciardi and Kipp, 2008). This suggests that overall efforts to reduce the importation or spread of all alien species should be helpful.

The Role of Biological Monitoring

The preceding sections illustrate the importance of biological data to understanding the complexities associated with urban and stormwater impacts to waterbodies. Although categories of urban stressors have been discussed individually, these stressors routinely, if not universally, co-occur in urban waterbodies. Their cumulative impacts are best measured with biological tools because the biota integrate the influence of all of these stressors.

Many programmatic aspects of the CWA arose as a response to rather obvious impacts of chemical pollutants that were occurring in surface waters during this time. The initial focus of water quality standards was on developing chemical criteria that could serve as engineering endpoints for waste treatment systems (e.g., NPDES permits). Rather general aquatic life goals for streams and rivers that were suitable for the initial focus of the CWA are now considered insufficient to deal with the complex suite of stressors limiting aquatic systems.
To that end, refined aquatic life goals and improved biological monitoring are essential for effective water quality management, including stormwater issues (NRC, 2001). Practical biological and physical monitoring tools have even been developed for very small headwater streams (Ohio EPA, 2002; Fritz et al., 2006), which are particularly affected by stormwater because of their prevalence (greater than 95 percent of channels), their relatively high surface-to-volume ratio, their role in nutrient and material processing, and their vulnerability to direct modification such as channelization and piping (Meyer and Wallace, 2001).

Surrogate indicators of stormwater impacts to aquatic life (such as TSS concentrations) have been widely used because direct biological measures were poorly developed and these surrogates were assumed to be important to pollutant delivery to urban streams. However, biological assessment has rapidly advanced in many states and can be readily applied or if needed modified to be sensitive to stormwater stressors (Barbour et al., 2008). As Karr and Chu (1999) warned, the management of complex systems requires measures that integrate multiple factors. Stormwater permitting is no different, and care must be taken to ensure that permitting and regulatory actions retain ecological relevance. Surrogate measures have an essential role in the assessment of individual SCMs; however, this needs to be kept in context with the entire suite of stressors likely to be important to the aquatic life goals in streams.

Stormwater management programs should not necessarily bear the burden of biological monitoring; rather, well-conceived biological monitoring should be the prelude of state and local government agencies (as discussed more extensively in Chapter 6). Refined aquatic life goals developed for all waters, including urban waters, measured with appropriate biological measures, should be the final endpoint for management. The collection of biological data needs to be closely integrated across multiple disciplines in order to be effective. Pomeroy et al. (2008) describe a multidisciplinary approach to study the effects of stormwater in urban settings, and Scholz and Booth (2001) also propose a monitoring approach for urban watersheds. Such efforts are not necessarily easy, and many institutions find pitfalls when trying to integrate scientific information across disciplines (Benda et al., 2002).

EPA water programs, such as the Total Maximum Daily Load (TMDL) program, have been criticized for having too narrow a focus on a limited number of traditional pollutants to the exclusion of important stressors such as hydrology, habitat alteration, and invasive taxa (Karr and Yoder, 2004)—all serious problems associated with stormwater and urbanization. The science has advanced significantly over the past decade so that biological assessment should be an essential tool for identifying stormwater impacts and informing the choice of SCMs in a region or watershed. Although biological responses to stressors in the ambient environment are by their nature correlative exercises, ecological epidemiology principles or “stressor identification” methods can identify likely causative agents of impairment with relatively high certainty in many instances (Suter, 1993, 2006; EPA, 2000). Coupled with other ambient and source moni-
EFFECTS OF URBANIZATION ON WATERSHEDS

Because monitoring information, biological information can form the basis for an effective stormwater program. As an example, Box 3-10 introduces the Impervious Cover Model (ICM), which was developed using correlative information on the association between impervious cover and biological metrics. The crux of the ICM is that stormwater management is tailored along a readily measurable gradient (impervious cover) that integrates multiple individual stressor categories that would otherwise be overlooked in the traditional pollutant-based approach to stormwater management. Even the form of the ICM (as conceptualized in Figure 3-43) matches that outlined for the BCG (Figure 3-38). Use of the ICM to improve the MS4 stormwater program is discussed in Chapter 6.

**Human Health Impacts**

Despite the unequivocal evidence of ecosystem consequences resulting from urban stormwater, a formal risk analysis of the human health effects associated with stormwater runoff is not yet possible. This is because (1) many of the most important waterborne pathogens have not been quantified in stormwater, (2) enumeration methods reported in the current literature are disparate and do not account for particle-bound pathogens, and (3) sampling times during storms have not been standardized nor are known to have occurred during periods of human exposure. Individual studies have investigated the runoff impacts on public health in freshwater (Calderon et al., 1991) and marine waters (Haile et al., 1999; Dwight et al., 2004; Colford et al., 2007). Although these studies provide ample evidence that stormwater runoff can serve as a vector of pathogens with potential health implications (for example, Ahn et al., 2005, found that fecal indicator bacteria concentrations could exceed California ocean bathing water standards by up to 500 percent in surf zones receiving stormwater runoff), it is difficult to draw conclusive inferences about the specific human health impacts from microbial contamination of stormwater. Calderon et al. (1991) concluded that the currently recommended bacterial indicators are ineffective for predicting potential health effects associated with water contaminated by non-point sources of fecal pollution. Furthermore, in a study conducted in Mission Bay, California, which analyzed bacterial indicators using traditional and non-traditional methods (chromogenic substrate and quantitative polymerase chain reaction), as well as a novel bacterial indicator and viruses, traditional fecal indicators were not associated with identified human health risks such as diarrhea and skin rash (Colford et al., 2007).

The Santa Monica Bay study (Haile et al., 1999) indicated that the risks of several health outcomes were higher for people who swam at storm-drain locations compared to those who swam farther from the drain. However, the list of health outcomes that were more statistically significant (fever, chills, ear discharge, cough and phlegm, and significant respiratory) did not include highly
The Impervious Cover Model (ICM) is a management tool that is useful for diagnosing the severity of future stream problems in a subwatershed. The ICM defines four categories of urban streams based on how much impervious cover exists in their subwatershed: high-quality streams, impacted streams, non-supporting streams, and urban drainage. The ICM is then used to develop specific quantitative or narrative predictions for stream indicators within each stream category (see Figure 3-43). These predictions define the severity of current stream impacts and the prospects for their future restoration. Predictions are made for five kinds of urban stream impacts: changes in stream hydrology, alteration of the stream corridor, stream habitat degradation, declining water quality, and loss of aquatic diversity.

The general predictions of the ICM are as follows. Stream segments with less than 10 percent impervious cover (IC) in their contributing drainage area continue to function as Sensitive Streams, and are generally able to retain their hydrologic function and support good-to-excellent aquatic diversity. Stream segments that have 10 to 25 percent IC in their contributing drainage area behave as Impacted Streams and show clear signs of declining stream health. Most indicators of stream health will fall in the fair range, although some segments may range from fair to good as riparian cover improves. The decline in stream quality is greatest toward the higher end of the IC range. Stream segments that range between 25 and 60 percent subwatershed impervious cover are classified as Non-Supporting Streams (i.e., no longer supporting their designated uses in terms of hydrology, channel stability habitat, water quality, or biological diversity). These stream segments become so degraded that any future stream restoration or riparian cover improvements are insufficient to fully recover stream function and diversity (i.e., the streams are so dominated by subwatershed IC that they cannot attain predevelopment conditions). Stream segments whose subwatersheds exceed 60 percent IC are physically altered so that they merely function as a conduit for flood waters. These streams are classified as Urban Drainage and consistently have poor water quality, highly unstable channels, and very poor habitat and biodiversity scores. In many cases, these urban stream segments are eliminated altogether by earthworks and/or storm-drain enclosure. Table 3-14 shows in greater detail how stream corridor indicators respond to greater subwatershed impervious cover.

### TABLE 3-14 General ICM Predictions Based on Urban Subwatershed Classification (CWP, 2004):

<table>
<thead>
<tr>
<th>Prediction</th>
<th>Impacted (IC 11 to 25%)</th>
<th>Non-supporting (IC 26 to 60%)</th>
<th>Urban Drainage (IC &gt; 60%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff as a Fraction of Annual Rainfall</td>
<td>10 to 20%</td>
<td>25 to 60%</td>
<td>60 to 90%</td>
</tr>
<tr>
<td>Frequency of Bankfull Flow per Year</td>
<td>1.5 to 3 per year</td>
<td>3 to 7 per year</td>
<td>7 to 10 per year</td>
</tr>
<tr>
<td>Fraction of Original Stream Network Remaining</td>
<td>60 to 90%</td>
<td>25 to 60%</td>
<td>10 to 30%</td>
</tr>
<tr>
<td>Fraction of Riparian Forest Buffer Intact</td>
<td>50 to 70%</td>
<td>30 to 60%</td>
<td>Less than 30%</td>
</tr>
<tr>
<td>Crossings per Stream Mile</td>
<td>1 to 2</td>
<td>2 to 10</td>
<td>None left</td>
</tr>
<tr>
<td>Ultimate Channel Enlargement Ratio</td>
<td>1.5 to 2.5 larger</td>
<td>2.5 to 6 times larger</td>
<td>6 to 12 times larger</td>
</tr>
<tr>
<td>Typical Stream Habitat Score</td>
<td>Fair, but variable</td>
<td>Consistently poor</td>
<td>Poor, often absent</td>
</tr>
<tr>
<td>Increased Stream Warming</td>
<td>2 to 4 °F</td>
<td>4 to 8 °F</td>
<td>8+ °F</td>
</tr>
<tr>
<td>Annual Nutrient Load</td>
<td>1 to 2 times higher</td>
<td>2 to 4 times higher</td>
<td>4 to 6 times higher</td>
</tr>
<tr>
<td>Wet Weather Violations of Bacteria Standards</td>
<td>Frequent</td>
<td>Continuous</td>
<td>Ubiquitous</td>
</tr>
<tr>
<td>Fish Advisories</td>
<td>Rare</td>
<td>Potential risk of accumulation</td>
<td>Should be presumed</td>
</tr>
<tr>
<td>Aquatic Insect Diversity</td>
<td>Fair to good</td>
<td>Fair</td>
<td>Very poor</td>
</tr>
<tr>
<td>Fish Diversity</td>
<td>Fair to good</td>
<td>Poor</td>
<td>Very poor</td>
</tr>
</tbody>
</table>

1. Based on annual storm runoff coefficient; ranges from 2 to 5% for undeveloped streams.
2. Predevelopment bankfull flood frequency is about 0.5 per year, or about one bankfull flood every two years.
3. Ultimate stream-channel cross-section compared to typical predevelopment channel cross section.
4. Typical increase in mean summer stream temperature in degrees Fahrenheit, compared with shaded rural stream.
5. Annual unit-area stormwater phosphorus and/or nitrogen load produced from a rural subwatershed.
6. As measured by benthic index of biotic integrity. Scores for rural streams range from good to very good.
7. As measured by fish index of biotic integrity. Scores for rural streams range from good to very good.
8. IC is not the strongest indicator of stream health below 10% IC, so the sensitive streams category is omitted from this table.

SOURCE: Adapted from Schueler (2004).
**Scientific Support for the ICM**

The ICM predicts that hydrological, habitat, water quality, and biotic indicators of stream health first begin to decline sharply at around 10 percent total IC in smaller catchments (Schueler, 1994). The ICM has since been extensively tested in ecoregions around the United States and elsewhere, with more than 200 different studies confirming the basic model for single stream indicators or groups of stream indicators (CWP, 2003; Schueler, 2004). Several recent research studies have reinforced the ICM as it is applied to first- to third-order streams (Coles et al., 2004; Homer et al., 2004; Deacon et al., 2005; Fitzpatrick et al., 2005; King et al., 2005; McBride and Booth, 2005; Cianfrina et al., 2006; Urban et al., 2006; Schueler et al., 2008).

Researchers have focused their efforts to define the specific thresholds where urban stream degradation first begins. There is robust debate as to whether there is a sharp initial threshold or merely a continuum of degradation as IC increases, although the latter is more favored. There is much less debate, however, about the dominant role of IC in defining the hydrologic, habitat, water quality, and biodiversity expectations for streams with higher levels of IC (15 to 60 percent).

**Caveats to the ICM**

The ICM is a powerful predictor of urban stream quality when used appropriately. The first caveat is that subwatershed IC is defined as total impervious area (TIA) and not effective impervious area (EIA). Second, the ICM should be restricted to first- to third-order alluvial streams with moderate gradient and no major point sources of pollutant discharge. The ICM is most useful in projecting the behavior of numerous stream health indicators, and it is not intended to be accurate for every individual stream indicator. In addition, management practices in the contributing catchment or subwatershed must not be poor (e.g., no deforestation, acid mine drainage, intensive row crops, etc.); just because a subwatershed has less than 10 percent IC does not automatically mean that it will have good or excellent stream quality if past catchment management practices were poor.

ICM predictions are general and may not apply to every stream within the proposed classifications. Urban streams are notoriously variable, and factors such as gradient, stream order, stream type, age of subwatershed development, and past land use can and will make some streams depart from these predictions. Indeed, these “outlier” streams are extremely interesting from the standpoint of restoration. In general, subwatershed IC causes a continuous but variable decline in most stream corridor indicators. Consequently, the severity of individual indicator impacts tends to be greater at the upper end of the IC range for each stream category.

**Effects of Catchment Treatment on the ICM**

Most studies that investigated the ICM were done in communities with some degree of catchment treatment (e.g., stormwater management or stream buffers). Detecting the effect of catchment treatment on the ICM involves a very complex and difficult paired watershed design. Very few catchments meet the criteria for either full treatment or the lack of it,
EFFECTS OF URBANIZATION ON WATERSHEDS

no two catchments are ever really identical, and individual catchments exhibit great variability from year to year. Not surprisingly, the first generation of research studies has produced ambiguous results. For example, seven research studies showed that ponds and wetlands are unable to prevent the degradation of aquatic life in downstream channels associated with higher levels of IC (Galli, 1990; Jones et al., 1996; Horner and May, 1999; Maxted, 1999; MNCPPC, 2000; Horner et al., 2001; Stribling et al., 2001). The primary reasons cited are stream warming (amplified by ponds), changes in organic matter processing, the increased runoff volumes delivered to downstream channels, and habitat degradation caused by channel enlargement.

Riparian forest cover is defined as canopy cover within 100 meters of the stream, and is measured as the percentage of the upstream network in this condition. Numerous researchers have evaluated the relative impact of riparian forest cover and IC on stream geomorphology, aquatic insects, fish assemblages, and various indices of biotic integrity. As a group, the studies suggest that indicator values for urban streams improve when riparian forest cover is retained over at least 50 to 75 percent of the length of the upstream network (Booth et al., 2002; Morley and Karr, 2002; Wang et al., 2003; Allan, 2004; Sweeney et al., 2004; Moore and Palmer, 2005; Cianfrina et al., 2006; Urban et al., 2006).

Application of the ICM to other Receiving Waters

Recent research has focused on the potential value of the ICM in predicting the future quality of receiving waters such as tidal coves, lakes, wetlands and small estuaries. The primary work on small estuaries by Holland et al. (2004) [references cited in CWP (2003), Lerberg et al. (2000)] indicates that adverse changes in physical, sediment, and water quality variables can be detected at 10 to 20 percent subwatershed IC, with a clear biological response observed in the range of 20 to 30 percent IC. The primary physical changes involve greater salinity fluctuations, greater sedimentation, and greater pollutant contamination of sediments. The biological response includes declines in diversity of benthic macroinvertebrates, shrimp, and finfish.

More recent work by King et al. (2005) reported a biological response for coastal plain streams at around 21 to 32 percent urban development (which is usually about twice as high as IC). The thresholds for important water quality indicators such as bacterial exceedances in shellfish beds and beaches appears to begin at about 10 percent subwatershed IC, with chronic violations observed at 20 percent IC (Mallin et al., 2001). Algal blooms and anoxia resulting from nutrient enrichment by stormwater runoff also are routinely noted at 10 to 20 percent subwatershed IC (Mallin et al., 2004).

The primary conclusion to be drawn from the existing science is that the ICM does apply to tidal coves and streams, but that the impervious levels associated with particular biological responses appear to be higher (20 to 30 percent IC for significant declines) than for freshwater streams, presumably due to their greater tidal mixing and inputs from nearshore ecosystems. The ICM may also apply to lakes (CWP, 2003) and freshwater wetlands (Wright et al., 2007) under carefully defined conditions. The initial conclusion is that the application of the ICM shows promise under special conditions, but more controlled research is needed to determine if IC (or other watershed metrics) is useful in forecasting receiving water quality conditions.

continues next page
Utility of the ICM in Urban Stream Classification and Watershed Management

The ICM is best used as an urban stream classification tool to set reasonable expectations for the range of likely stream quality indicators (e.g., physical, hydrologic, water quality, habitat, and biological diversity) over broad ranges of subwatershed IC. In particular, it helps define general thresholds where water quality standards or biological narrative conditions cannot be consistently met during wet weather conditions (see Table 6-2). These predictions help stormwater managers and regulators to devise appropriate and geographically explicit stormwater management and subwatershed restoration strategies for their catchments as part of MS4 permit compliance. More specifically, assuming that local monitoring data are available to confirm the general predictions of the ICM, it enables managers to manage stormwater within the context of current and future watershed conditions.

Credible gastrointestinal illness, which is curious because the vast majority of epidemiological studies worldwide suggest a causal dose-related relationship between gastrointestinal symptoms and recreational water quality measured by bacterial indicator counts (Pruss, 1998). Dwight et al. (2004) found that surfers in an urban environment reported more symptoms than their rural counterparts; however, water quality was not specifically evaluated in that study.

To better assess the relationship between swimming in waters contaminated by stormwater, which have not been influenced by human sewage, and the risk of related illness, the California Water Boards and the City of Dana Point have initiated an epidemiological study. This study will be conducted at Doheny Beach, Orange County, California, which is a beach known to have high fecal indicator bacteria concentrations with no known human source. The project will examine new techniques for measuring traditional fecal indicator bacteria, new species of bacteria, and viruses to determine whether they yield a better relationship to human health outcomes than the indicators presently used in California. The study is expected to be completed in 2010. In addition, the State of California is researching new methods for rapid detection of beach bacterial indicators and ways to bring these methods into regular use by the environmental monitoring and public health communities to better protect human health.

CONCLUSIONS AND RECOMMENDATIONS

The present state of the science of stormwater reflects both the strengths and weaknesses of historic, monodisciplinary investigations. Each of the component disciplines—hydrology, geomorphology, aquatic chemistry, ecology, land use, and population dynamics—have well-tested theoretical foundations and useful predictive models. In particular, there are many correlative studies showing how parameters co-vary in important but complex and poorly understood ways (e.g., changes in fish community associated with watershed road
density or the percentage of IC). Nonetheless, efforts to create mechanistic links between population growth, land-use change, hydrologic alteration, geomorphic adjustments, chemical contamination in stormwater, disrupted energy flows, and biotic interactions, to changes in ecological communities are still in development. Despite this assessment, there are a number of overarching truths that remain poorly integrated into stormwater management decision making, although they have been robustly characterized and have a strong scientific basis. These are expanded upon below.

There is a direct relationship between land cover and the biological condition of downstream receiving waters. The possibility for the highest levels of aquatic biological condition exists only with very light urban transformation of the landscape. Even then, alterations to biological communities have been documented at such low levels of imperviousness, typically associated with roads and the clearing of native vegetation, that there has been no real “urban development” at all. Conversely, the lowest levels of biological condition are inevitable with extensive urban transformation of the landscape, commonly seen after conversion of about one-third to one-half of a contributing watershed into impervious area. Although not every degraded waterbody is a product of intense urban development, all highly urban watersheds produce severely degraded receiving waters. Because of the close and, to date, inexorable linkage between land cover and the health of downstream waters, stormwater management is an unavoidable offshoot of watershed-based land-use planning (or, more commonly, its absence).

The protection of aquatic life in urban streams requires an approach that incorporates all stressors. Urban Stream Syndrome reflects a multitude of effects caused by altered hydrology in urban streams, altered habitat, and polluted runoff. Focusing on only one of these factors is not an effective management strategy. For example, even without noticeably elevated pollutant concentrations in receiving waters, alterations in their hydrologic regimes are associated with impaired biological condition. Achieving the articulated goals for stormwater management under the CWA will require a balanced approach that incorporates hydrology, water quality, and habitat considerations.

The full distribution and sequence of flows (i.e., the flow regime) should be taken into consideration when assessing the impacts of stormwater on streams. Permanently increased stormwater volume is only one aspect of an urban-altered storm hydrograph. It contributes to high in-stream velocities, which in turn increase streambank erosion and accompanying sediment pollution of surface water. Other hydrologic changes, however, include changes in the sequence and frequency of high flows, the rate of rise and fall of the hydrograph, and the season of the year in which high flows can occur. These all can affect both the physical and biological conditions of streams, lakes, and wetlands. Thus, effective hydrologic mitigation for urban development cannot just
A single design storm cannot adequately capture the variability of rain and how that translates into runoff or pollutant loadings, and thus is not suitable for addressing the multiple objectives of stormwater management. Of particular importance to the types of problems associated with urbanization is the size of rain events. The largest and most infrequent rains cause near-bank-full conditions and may be most responsible for habitat destruction; these are the traditional “design storms” used to design safe drainage systems. However, moderate-sized rains are more likely to be associated with most of the annual mass discharges of stormwater pollutants, and these can be very important to the eutrophication of lakes and nearshore waters. Water quality standards for bacterial indicators and total recoverable heavy metals are exceeded for almost every rain in urban areas. Therefore, the whole distribution of storm size needs to be evaluated for most urban receiving waters because many of these problems co-exist.

Roads and parking lots can be the most significant type of land cover with respect to stormwater. They constitute as much as 70 percent of total impervious cover in ultra-urban landscapes, and as much as 80 percent of the directly connected impervious cover. Roads tend to capture and export more stormwater pollutants than other land covers in these highly impervious areas because of their close proximity to the variety of pollutants associated with automobiles. This is especially true in areas of the country having mostly small rainfall events (as in the Pacific Northwest). As rainfall amounts become larger, pervious areas in most residential land uses become more significant sources of runoff, sediment, nutrients, and landscaping chemicals. In all cases, directly connected impervious surfaces (roads, parking lots, and roofs that are directly connected to the drainage system) produce the first runoff observed at a storm-drain inlet and outfall because their travel times are the quickest.

Generally, the quality of stormwater from urbanized areas is well characterized, with the common pollutants being sediment, metals, bacteria, nutrients, pesticides, trash, and polycyclic aromatic hydrocarbons. These results come from many thousands of storm events from across the nation, systematically compiled and widely accessible; they form a robust data set of utility to theoreticians and practitioners alike. These data make it possible to accurately estimate pollutant concentrations, which have been shown to vary by land cover and by region across the country. However, characterization data are relatively sparse for individual industrial operations, which makes these sources less amenable to generalized approaches based on reliable assumptions of pollutant types and loads. In addition, industrial operations vary greatly from site to site, such that it may be necessary to separate them into different categories in order to better understand industrial stormwater quality.
Nontraditional sources of stormwater pollution must be taken into consideration when assessing the overall impact of urbanization on receiving waterbodies. These nontraditional sources include atmospheric deposition, snowmelt, and dry weather discharges, which can constitute a significant portion of annual pollutant loadings from storm systems in urban areas (such as metals in Los Angeles). For example, atmospheric deposition of metals is a very significant component of contaminant loading to waterbodies in the Los Angeles region relative to other point and nonpoint sources. Similarly, much of the sediment found in receiving waters following watershed urbanization can come from streambank erosion as opposed to being contributed by polluted stormwater.

Biological monitoring of waterbodies is critical to better understanding the cumulative impacts of urbanization on stream condition. Over 25 years ago, individual states developed the concept of regional reference sites and developed multi-metric indices to identify and characterize degraded aquatic assemblages in urban streams. Biological assessments respond to the range of non-chemical stressors identified as being important in urban waterways including habitat degradation, hydrological alterations, and sediment and siltation impacts, as well as to the influence of nutrients and other chemical stressors where chemical criteria do not exist or where their effects are difficult to measure directly (e.g., episodic stressors). The increase in biological monitoring has also helped to frame issues related to exotic species, which are locally of critical importance but completely unrecognized by traditional physical monitoring programs.

Epidemiological studies on the human health risks of swimming in freshwater and marine waters contaminated by urban stormwater discharges in temperate and warm climates are needed. Unlike with aquatic organisms, there is little information on the health risks of urban stormwater to humans. Standardized watershed assessment methods to identify the sources of human pathogens and indicator organisms in receiving waters need to be developed, especially for those waters with a contact-recreation use designation that have had multiple exceedances of pathogen or indicator criteria in a relatively short period of time. Given their difficulty and expense, epidemiological studies should be undertaken only after careful characterization of water quality and stormwater flows in the study area.

REFERENCES


Brantley, A. S., and T. G. Townsend. 1999. Leaching of pollutants from re-


EFFECTS OF URBANIZATION ON WATERSHEDS


Deacon, J., S. Soule, and T. Smith. 2005. Effects of urbanization on stream quality at selected sites in the seacoast region in New Hampshire, 2001-


EFFECTS OF URBANIZATION ON WATERSHEDS

Meador (eds.). American Fisheries Society, Symposium 47.


LAC DPW (Los Angeles County, Department of Public Works). 1985. Plan for Flood Control and Water Conservation, Maps 2, 3, 7, 16, and 17. LAC DPW, Los Angeles, CA.


Maxted, J. R. 1999. The effectiveness of retention basins to protect aquatic life and physical habitat in three regions of the United States. Pp. 215–222 In:
EFFECTS OF URBANIZATION ON WATERSHEDS


Novotny, V., et al. 1986. Effect of Pollution from Snow and Ice on Quality of
EFFECTS OF URBANIZATION ON WATERSHEDS

Oliver, B., J. B. Milne, and N. LaBarre. 1974. Chloride and lead in urban snow. Journal of Water Pollution Control Federation 46(4).
Lewis Publishers.
EFFECTS OF URBANIZATION ON WATERSHEDS  249


Rogge, W. F., L. M. Hildemann, M. A. Mazeuk, G. R. Cass, and B. R. T. Si-
Rosselot, K. S. 2006b. Copper Released from Non-Brake Sources in the San Francisco Bay Area, Final Report, Brake Pad Partnership, January.


SMBRP (Santa Monica Bay Restoration Project). 1994. Characterization Study
of the Santa Monica Bay Restoration Plan. Santa Monica Bay Restoration Project, Monterey Park, CA.


Suter, G. 2006. Ecological risk assessment and ecological epidemiology for
Walters, D. M., M. C. Freeman, D. S. Leigh, B. J. Freeman, and C. M. Pringle.


Wells, C. 1995. Skinny streets and one-sided sidewalks: a strategy for not pavi
}

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